Assessing the Success of Restoration Plantings at Cape Foulwind, New Zealand

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ABSTRACT

Holcim (New Zealand) Ltd. operate a quarry near Cape Foulwind, New Zealand. Quarrying operations have a dramatic effect on the environment. Consequently, the company has developed a restoration strategy that aims to mitigate the environmental and visual impacts of quarry operations. The objective of this study was to determine how successful the restoration plantings at Cape Foulwind have been to date. Achieving restoration success is dependent upon meeting the goals established for the restoration project. The specificity, appropriateness, and ease of measurement of these goals play a large part in determining the level to which restoration plantings can be deemed successful.

The six planted restoration study sites investigated (planted 3 to 22 years prior to this study) were compared with three forest remnant sites, acting as a reference. Determination of the level of restoration success involved investigation of both ecosystem structure and functional attributes. Vegetation composition, ground active invertebrates and various ecosystem attributes, including soil, litter depth and decomposition, and seed rain were investigated using numerous diversity indices and ordination techniques where appropriate.

The results of this study suggest that while complete success of these restoration plantings has not yet occurred, attributes necessary for initial success were present. Planted restoration sites were facilitating the entry of novel regenerating species. The current limiting factor to progression within the planted restoration study sites appears to be the lack of full canopy cover, and subsequent development of suitable microclimatic conditions. A large difference was apparent in composition and abundance of ground active invertebrate communities in planted restoration and remnant study sites. Of the environmental variables investigated, litter depth was found to be the key driver of invertebrate distribution over the nine study sites.

Holcim's restoration plantings at Cape Foulwind have successfully provided new habitat for native biodiversity, while facilitating development of ecosystem structure and functioning. Importantly, they are increasing the connectivity between the native forest remnants that are present, enhancing the aesthetic appeal of the quarry area.

1. INTRODUCTION

Holcim (New Zealand) Ltd., hereafter referred to as Holcim, operate a quarry near Cape Foulwind, New Zealand. This quarry supplies raw materials necessary for cement manufacturing; the primary component, limestone, and the secondary component, marl. The quarry proper and surrounding land owned by Holcim occupies an area of approximately 150 ha (Holcim, 2002). Quarrying operations have a major effect on the surrounding environment. Consequently, Holcim have undertaken a restoration project for the quarry proper and surrounding areas to mitigate the environmental and visual impacts of quarry operations (Norton, 1992). This restoration effort has thus far involved the planting of native species to restore the quarry surrounds to a fully functioning, self-sustaining ecosystem. The overall goal of this research project was to determine the initial success of restoration plantings surrounding the quarry proper.

This chapter will provide background information on the topic of ecological restoration, including justification for restoration, the necessity of goals in restoration ecology, an overview of the specific goals of Holcim and a brief discussion of appropriate measures of success for ecological restoration projects. Following this summary, objectives of this research project and a brief outline of the structure of this thesis is detailed.

1.1. What is Ecological Restoration?

A frequently cited definition of ecological restoration comes from a report issued by the U.S. National Research Council (1992, p.18), where restoration is defined as "the return of an ecosystem to a close approximation of its condition prior to disturbance" (Bradshaw, 2002). Atkinson (1988, p.1) describes ecological restoration as "active intervention and management to restore or partially restore biotic communities, both plants and animals, as fully functioning systems". Norton (1995) extends this definition further, to include the restoration of the physical environment associated with plants and animals, as fully functioning and sustainable systems with a predominance of indigenous species. Restoration is defined in the New Oxford

Dictionary of English (1998) as the action of returning something to a former condition.

A more general definition of ecological restoration is provided by Jackson et al (1995). They describe ecological restoration as the process of repairing damage caused by humans to the dynamics and diversity of indigenous ecosystems. Such a definition does not necessitate the return of a system to a one particular ideal state, but allows for a broader approach to restoration efforts. A similar approach to defining ecological restoration was taken by the Society for Ecological Restoration who defines ecological restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed" (SER, 2003).

As is apparent, much discussion can be found in the restoration ecology literature regarding appropriate definitions for ecological restoration. Most definitions focus on re-establishing what might have occurred on a site had it remained undisturbed (Hobbs & Norton, 1996), thereby implying a single desirable state exists for restoration. However, this insinuation may restrict restoration efforts. It is unrealistic to assume that an ecosystem can be returned to its former condition, incorporating all original elements and functions (Saunders et al, 1993). When restoration aims to faithfully return what was at a site, it can be difficult to define an appropriate reference site (Bradshaw, 2002). Should restoration aim to return what was there prior to damage to the ecosystem occurred, or prior to when humans started to modify it? Is an appropriate reference system the ecosystem as it occurred in pre-European times (1840 AD in New Zealand), or pre-Polynesian times (1200 AD)? Ecological restoration that is concerned with returning a site to some previous state is fraught with difficulties, and is viewed as naïve, difficult, or impossible for theoretical and pragmatic reasons (Norton, 1995; Cowell, 1993).

Definitions that focus on returning a site to a particular form can impose a static view of ecosystems when ecosystems are dynamic entities; restoration efforts should be applied with this in mind (Hobbs & Norton, 1996). Ecosystems are complex dynamic entities, which are continually changing (Higgs, 1997; Pickett & Parker, 1994). If a site remained undisturbed it would have changed with time anyway, therefore using

only an historical reference is not realistic. It is inappropriate and indefeasible to recreate a single past 'natural' condition, due to extinctions and invasions (Norton, 1995). Such an idea of naturalness can cause difficulties, as it infers that disturbances resulting from human actions are different from non-anthropocentric disturbances (Callicott et al, 1998). If the reference state chosen is the condition prior to when humans or Europeans first encountered it, this fails to place humans in the long-term development of the ecosystem once the appropriate structure and function have been achieved (Cowell, 1993). Recognition that many alternative states are possible for ecosystems in any location, even under natural conditions, provides reason to reconsider the basis for the use of a single reference system (Hobbs & Norton, 1996). Cowell (1993) appropriately suggests that a past condition should not be used as a goal for restoration, but as a reference point for future development.

The use of one particular natural state against which restoration efforts are measured may lead to the setting of unrealistic, ambiguous goals, because detailed knowledge of the composition, structure and functions of historic systems are rarely known, making it difficult to measure success (Hobbs & Norton, 1996). Definition of reference systems can be difficult as nature is variable both spatially and temporally (White & Walker, 1997).

The practice of ecological restoration aims to accelerate ecosystems along a desired trajectory to a reference ecosystem by accelerating succession (Honnay et al, 2002). Parker & Pickett (1997) argue that restoration should be considered as a process, with the appropriate degree of intervention determined by particular contextual circumstances (Bradshaw, 2002). Knowledge of the environmental context of a restoration project and the historical context of a site is essential for the undertaking of ecological restoration (Kettle et al, 2000).

Restoration is a process, occurring along a continuum from the conservation of a single species within relatively intact ecosystems, to the building of ecosystems from bare ground. The position of restoration projects along the continuum will depend on the extent of damage, from limited management of relatively unmodified sites, to the rebuilding of entirely degraded sites, such as mining areas (Montalvo et al, 1997;

Hobbs & Norton, 1996). Atkinson (1988) specifies three aspects involved in all restoration projects: (1) the definition of a restoration goal; defined in terms of community composition; (2) active intervention to restore plants and / or animals formerly present; and (3) monitoring of progress, with further intervention when necessary. Similarly, Jackson et al, (1995) state that restoration projects will inevitably require: (1) judgement of need; (2) an ecological approach; and (3) recognition of the limitations of ecological restoration.

There is much discussion in restoration ecology literature regarding appropriate definitions for ecological restoration. To further complicate the issue, numerous terms are prevalent throughout the literature, such as restoration, rehabilitation, reallocation, and reclamation, to name a few. Hobbs & Norton (1996) suggest that arguing over semantics in this way can divert restoration ecologists from the task at hand. Bradshaw (2002) argues that it would be useful if restoration was used as a blanket term to describe all activities that seek to upgrade damaged land to a form in which biological potential is restored. This, he suggests, would involve attending to all ecosystem characteristics. Use of the term 'ecological', restoration can be limited to restoring individual components of an ecosystem, rather than its entirety. The use of the word restoration encourages the consideration of all fundamental processes necessary for an ecosystem to function, as well as the importance of natural processes in restoration, particularly those involved in succession (Bradshaw, 2002).

Goal definition can be seen as more important than the definition of specific terms such as restoration. It is essential to a restoration project's success, providing a guideline along which to proceed (Hobbs & Harris, 2001).

1.2. Justification for Ecological Restoration

Humans have substantially modified the environment in the past and continue to have dramatic adverse effects. Although the option exists to take a passive approach (i.e. do nothing) letting systems repair themselves, natural processes take extensive periods of time, decades or centuries (Dobson et al, 1997), and degraded ecosystems may persist as scars on landscapes (Robinson & Handel, 1993). It can alternatively be argued that we have a moral obligation to assist in repairing ecosystems that we have degraded.

Ecological restoration is an obvious choice to manipulate successional changes and speed up ecosystem recovery (Palmer et al, 1997). Ecological restoration provides a redemptive opportunity to reverse some of the detrimental effects that humans have had on the environment (Dobson et al, 1997; Higgs, 1997).

In New Zealand, site restoration is a general requirement of mining, intended for companies to make amends for environmental damaged caused by their activities. Section 17(1) of the Resource Management Act 1991 states that:

> *Every person has a duty to avoid, remedy, or mitigate any adverse effect on the environment arising from an activity carried on by or on behalf of that person, whether or not the activity is in accordance with a rule in a plan, a resource consent, section 10 (certain existing uses protected), or section 20 (certain existing lawful activities allowed).*

Holcim began mining at Cape Foulwind prior to the Resource Management Act (1991) coming into effect. As such, Holcim were not required by law to restore the area. Instead, restoration efforts were initially undertaken primarily as a public relations endeavour by the company, to improve the visual impact of the landscape, with the additional benefit of mitigating the negative environmental effects of the mining operations.

A variety of motives can be used to justify the importance of ecological restoration. Restoration projects can be used as an educational resource, so that people can learn about indigenous biotic communities (Atkinson, 1988). Restoration provides an accessible way in which local communities can become involved in nature conservation, see results of restoration and witness ecosystems developing over time (Norton, 1995). At present, our scientific knowledge of the workings of ecosystems is by no means complete, and can be enhanced by knowledge and questions brought to the forefront by ecological restoration (Cairns & Heckman, 1996). Restoration projects offer a feedback loop to ecology, allowing hypotheses to be tested (Webb, 1996). Dobson et al (1997) suggest that ecological restoration provides the opportunity for people to test their knowledge and ideas, and provide insights regarding assembly rules for communities and the way in which ecosystems function. As such, restoration projects can be used as an 'acid test' for our ecological knowledge (Bradshaw, 1983). Further, ecological restoration provides the opportunity

for genetic variation to be conserved (Atkinson, 1988) through the use of ecologically appropriate species (Norton, 1995). The site of a restoration project can be used as a potential source of native flora and fauna, to restore neighbouring areas. Restoration can provide buffers that will link previously isolated fragments and alleviate the effects of external factors such as edge effects (Hobbs & Norton, 1996). Finally, aesthetic benefits resulting from ecological restoration should not be underestimated (Atkinson, 1988).

1.3. Goals of Ecological Restoration

Establishing goals at the outset of a restoration project is vital as goals establish expectations, drive plans and determine the extent of monitoring necessary (Ehrenfeld, 2000). Goals for ecological restoration will be highly variable, as will techniques used to repair the system, depending on the severity of ecosystem degradation (Ehrenfeld, 2000; Reay & Norton, 1999a; Hobbs & Norton, 1996). Goals need to be appropriate and specific to individual restoration projects and relevant in terms of the scope and reasons for the restoration effort (Ehrenfeld, 2000). Goals appropriate for increasing the aesthetic qualities of the landscape will no doubt vary considerably from goals for a restoration project aiming to restore an ecosystem to a self-sustaining, fully functioning ecosystem (Montalvo et al, 1997; Hobbs & Norton, 1996). When the goal is to restore a self-sustaining ecosystem, this has long-term implications; therefore needs to be considered at a larger scale (both spatially and temporally) (Parker, 1997). If goals are limited to initiating a certain species composition, then a focus on biological interactions is appropriate (Parker, 1997).

Reference information is frequently used to define goals of restoration projects. Such reference sites are useful to determine the potential of sites to be restored and to evaluate the success of restoration efforts (White & Walker, 1997). The importance of reference sites is emphasised by Aronson et al (1995). Reference sites should be chosen prior to the commencement of restoration efforts. White & Walker (1997) suggest that reference information applies to a particular reference site at a particular time (i.e. spatially and temporally based). At the minimum, a reference site may provide sufficient inspiration and motivation to keep restoration projects moving (Aronson et al, 1995). An alternative to using a past system as a model for restoration

is to use an extant system as a reference. However, Hobbs & Harris (2001) note that this approach is not without its difficulties, as systems must be appropriately matched.

Despite the apparent importance of a reference system, care needs to be taken to ensure that resultant goals are not unattainable due to emphasis on restoring to a static condition (Hobbs & Harris, 2001), as discussed in Section 1.1. Goals need to be dynamic and take into account the variable nature of the environment (Hobbs & Norton, 1996). Indeed, restoration goals should recognise alternative states that a natural system may adopt (Westman, 1991). Recognition of restoration as a process requires more than one reference state to be considered (Webb, 1996). Westman (1991) suggests that in order to achieve a working definition of a restoration goal, it is necessary to choose structural and functional parameters that will serve as criteria of success. An ecosystem approach to restoration conserves and manages both structure and function of an entire ecosystem rather than managing individual organisms of interest (Lapin & Barnes, 1995). In recognising the dynamic nature of ecosystems, it may be preferential to establish a series of short- and long-term goals for restoration projects (Hobbs & Harris, 2001).

Hobbs & Harris (2001) suggest that a clear rationale for goal-setting is necessary, which takes into account the nature of the system being restored, factors leading to degradation, and the type of actions required to achieve restoration of various ecosystem attributes. Numerous factors may be considered when setting goals for ecological restoration (Hobbs & Harris, 2001). Hobbs & Norton (1996) set out various attributes that might be considered; ecosystem composition, structure, function, heterogeneity and resilience. Higgs (1997) suggested that 'ecological fidelity' could form the focus of restoration goals. Ecological fidelity is comprised of three elements: structural / compositional replication, functional success and durability (Hobbs & Harris, 2001).

Prior to the commencement of restoration activities, a rigorous assessment of the current state of the system in question needs to occur, in addition to considering factors that have lead to that state (Hobbs & Harris, 2001; Hobbs & Norton, 1996).

Without the establishment of appropriate goals, monitoring (evaluating performance in relation to goals) and evaluation of success will not be possible (Westman, 1991). Restoration success depends on the arrival at mutually agreed upon goals via open and effective processes (Hobbs & Harris, 2001; Higgs, 1997). In addition to goals being established at the outset of restoration projects, restoration goals should be rigorously monitored, which may need to continue for a considerable period of time (Holmes & Richardson, 1999). Such monitoring processes will enable recognition of success to be achieved more readily by providing benchmarks for evaluation (Jackson et al, 1995), with the added benefit of potentially identifying gaps in ecologists' understanding, which will assist in directing future research efforts (Holmes & Richardson, 1999).

1.3.1. Goals of Holcim (New Zealand) Ltd.

The overall goal of Holcim's restoration management programme is to restore a mosaic of indigenous forest and wetland communities similar to that which would have existed prior to human (principally European) arrival (Norton, 1992). Areas that are to be restored include farmland, the quarry operations area and overburden dumps. The excavation process has left a substantial pit, which after the completion of mining will be filled to form a lake. From this lake, it is envisioned that there will be wetlands grading back into indigenous forest.

Holcim's restoration philosophy consists of three main principles or tenants. The first is to create a self-sustaining ecosystem, in comparison to a garden requiring high levels of maintenance. The aim is for minimum human intervention after planting, by integrating management with the natural processes that lead to vegetation and forest development. Mechanisms necessary to achieve the desired outcome for this restoration project exist naturally in nature, it follows that restoration should mimic natural succession, which is the second tenant of Holcim's restoration philosophy. This implies that vegetation types used to encourage forest growth will be appropriate early successional species. The third tenant is to adopt an adaptive management approach, which is pragmatic for the company, trying to start restoration early and learn as it progresses, as opposed to trialling restoration off site (Norton, 1992).

The restoration plan for the quarry and its surrounds has been divided into four restoration zones based on landform position and mining impacts: (1) coastal restoration zone (primarily farmland adjacent to Tauranga Bay), (2) wetland restoration zone (planned lake and associated wetlands), (3) quarry restoration zone (areas of workings not flooded and quarry slopes), and (4) inland restoration zone (land adjacent to Tauranga Bay Road and inland farmland). This project is focused on assessing the success of plantings of various ages in the coastal and inland restoration zones (zones 1 & 4).

An additional consideration in the restoration of the quarry area is the close proximity of the quarry to public conservation land at Cape Foulwind, one of the West Coast's most popular tourist destinations. There stands the potential for the restored area to be combined with conservation and recreational areas at Cape Foulwind and Tauranga Bay to enhance overall ecological and scenic values at Tauranga Bay (Norton, 1992). This implies there is a visual component involved in the goals of restoration, mitigating the visual effects of mining, as well as underlying implications that indigenous wildlife such as bird species will be enhanced. Thus, social as well as ecological goals of restoration management are equally applicable. However, the initial success of this restoration project investigated within this thesis is limited to ecological success.

1.4. Success of Ecological Restoration

The success of restoration projects and issues related to effective measures of success will be discussed in more detail at the beginning of Chapter 6. However, it is important to emphasise at the outset the importance of using both structural and functional measures of restoration success. The aim of restoration is to restore a fully functioning, self-sustaining system that is integrated with the landscape within which it occurs (Bradshaw, 2002). Numerous suggestions of appropriate measures of success have been offered (e.g. Jackson et al, 1995; Norton, 1995; Aronson et al, 1993a; Aronson et al, 1993b; Cairns, 1991). However, within all appropriate measures of success, consideration of both structure and function is necessary. Re-creating the physical form of an ecosystem, without restoring naturally occurring ecological functions, or re-creating functions in a system, which bears little resemblance to a

natural ecosystem, does not constitute complete restoration (Bradshaw, 2002; Berger, 1993; U.S. National Research Council, 1992). It has previously been assumed that restoration of ecosystem structure will achieve restoration of ecosystem function. However, structural and functional attributes often develop at independent rates, casting the validity of this assumption into question (Westman, 1991).

Similar to the variation in goals appropriate for different ecological restoration projects, criteria used to judge success vary also (Montalvo et al, 1997). Reay & Norton (1999a) suggest that success occurs along a continuum from the establishment of initial plantings through to the formation of a self-sustaining, fully functioning ecosystem. Readily measurable indicators of ecosystem function need to be chosen (litter decomposition, seed dispersal and regeneration have been chosen for the purposes of this research) to allow levels of success to be determined.

1.5. Objectives

The overall objective of this research project was to assess the initial success of the restoration plantings at the Holcim quarry based at Cape Foulwind, New Zealand. In order to achieve this objective an investigation into various aspects of the development of the restoration plantings thus far, was undertaken.

Specifically to:

1. evaluate whether a progression in the development of vascular plant and invertebrate species composition has occurred with increasing time since planting;
2. determine whether the ecosystem processes of litter decomposition and seed dispersal have been established within the planted restoration study sites;
3. define a measure of success for ecological restoration and then determine whether ecological restoration within the study area has been successful;
4. draw inferences on the possible future development of the restoration plantings; and
5. comment on the management implications of this research.

1.6. Thesis Outline

Chapter 2: Study Area

Chapter 2 provides an overview of the study area, including geology, soils, climate and vegetation pattern and history. It also contains a description of the nine study sites used for this research.

Chapter 3: Vegetation Composition

The vegetation composition chapter investigates the diversity of vegetation present in each of the nine study sites. It attempts to provide insight into the ability of the restoration plantings to facilitate regeneration. Only a preliminary discussion of the vegetation composition results is presented within this chapter. These results are placed within the context of the rest of the study and evaluation of restoration success in Chapter 6.

Chapter 4: Ground Active Invertebrates

Chapter 4 provides information on the diversity, abundance and distribution of ground invertebrate communities detected within the nine study sites. Similar to Chapter 3, discussion of results is limited to the results contained within this chapter, with a further discussion in Chapter 6.

Chapter 5: Ecosystem Attributes

Chapter 5 covers a broad range of ecosystem attributes, investigating soil properties, seed rain, ground litter, and light data. A brief discussion of pertinent results of this chapter occurs at the end, although, like the previous two chapters, placing results found here in the context of restoration success occurs in Chapter 6.

Chapter 6: Discussion

A measure of restoration success is defined in this discussion chapter, with the integration of the results from chapters 3, 4, and 5. Furthermore, this chapter determines the level of initial success achieved within planted restoration study sites. Management implications resulting from this study are also discussed.

2. STUDY AREA

2.1. Introduction

The purpose of this chapter is to provide a brief overview of the general study area, including geology, soils, climate and vegetation pattern and history. Brief descriptions of each of the nine study sites selected are also presented, including restoration history.

2.2. Landform and Geology

Cape Foulwind lies approximately 16 km from the township of Westport and is located within the Foulwind Ecological District (Figure 2.1; Figure 2.2), with latitude of 41° 45.6′ S and longitude of 171° 28.0′ E. The Foulwind Ecological District is bounded to the north and west by the Tasman Sea, to the east by the Buller River, and to the southeast by the Paparoa Range. The area is comprised of a series of gently sloping marine terraces, separated by short steep scarps, east of SH 6 (Norton, 1992). The marine terraces descend from 64 m a.s.l. at the Cape to a low point (20 m) about halfway along Wilsons Lead Road, then rise to the highest point, about 200 m a.s.l., on Caroline Terrace (Mew & Ross, 1991). These marine surfaces are in various stages of dissection, overtopped by old, migrated dunes and cut into, in part, by flat-bottomed scour channels and former lagoons. Landforms present are the result of Tertiary and Quaternary deposits overlying an eroded Precambrian metamorphic complex, which contains granites of Palaeozoic age. Outcrops of the pre-Quaternary rocks are limited to coastal cliffs at Cape Foulwind (Norton, 1992). Up against the mountains of the Paparoa Range are some gently sloping fans, particularly on Caroline Terrace (Mew & Ross, 1991). Small valleys extend into terraces from the coast, where rivers have previously been active. A narrow coastal plain exists near Carters Beach and a lagoon system occurs at the mouth of the Okari River (Norton, 1992).

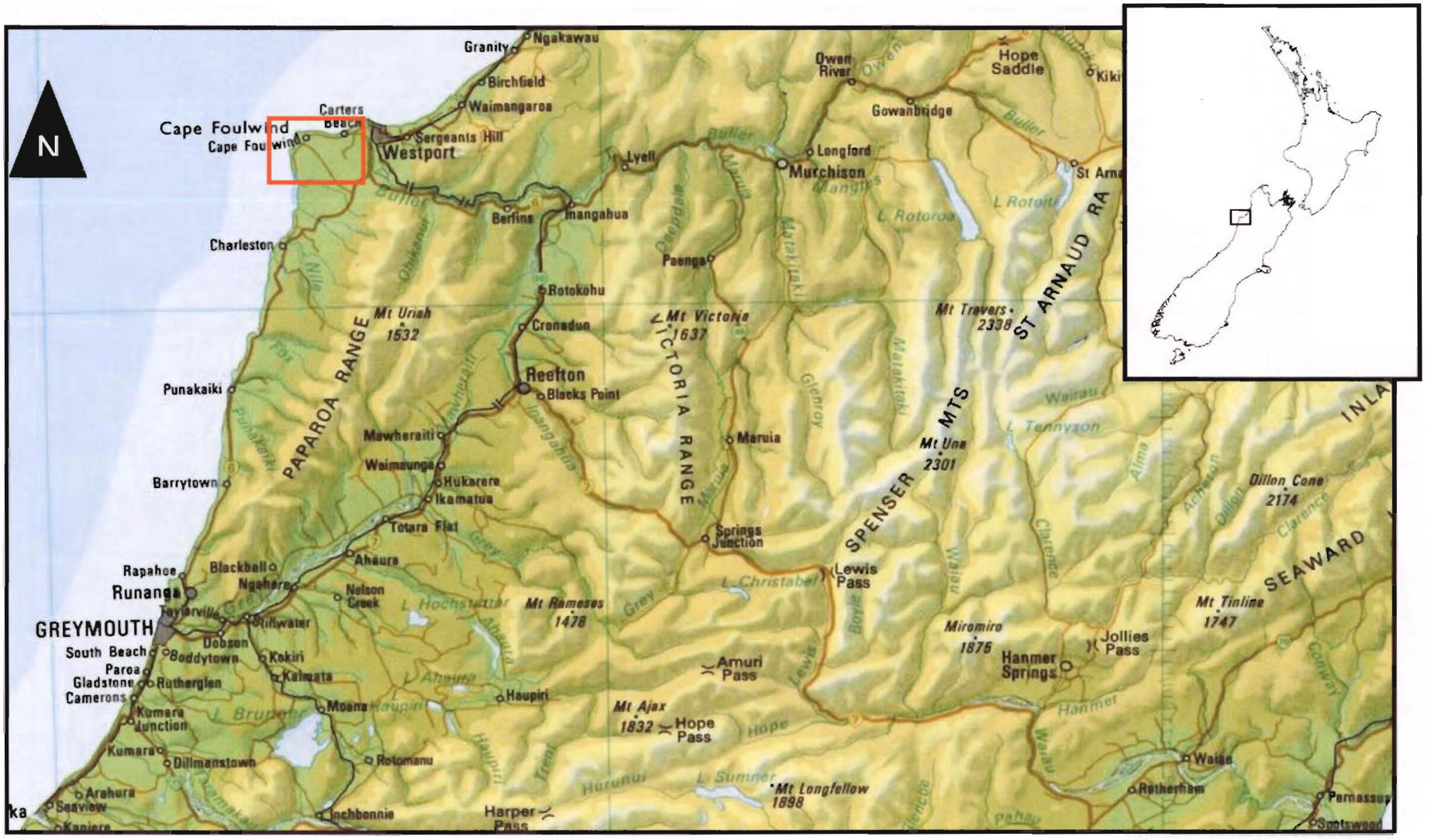


Figure 2.1: Location of the study area (indicated by red outline), Cape Foulwind, West Coast, South Island, New Zealand (Map taken from Dept. of Survey and Land Information, Infomap 242-3 North meets South).

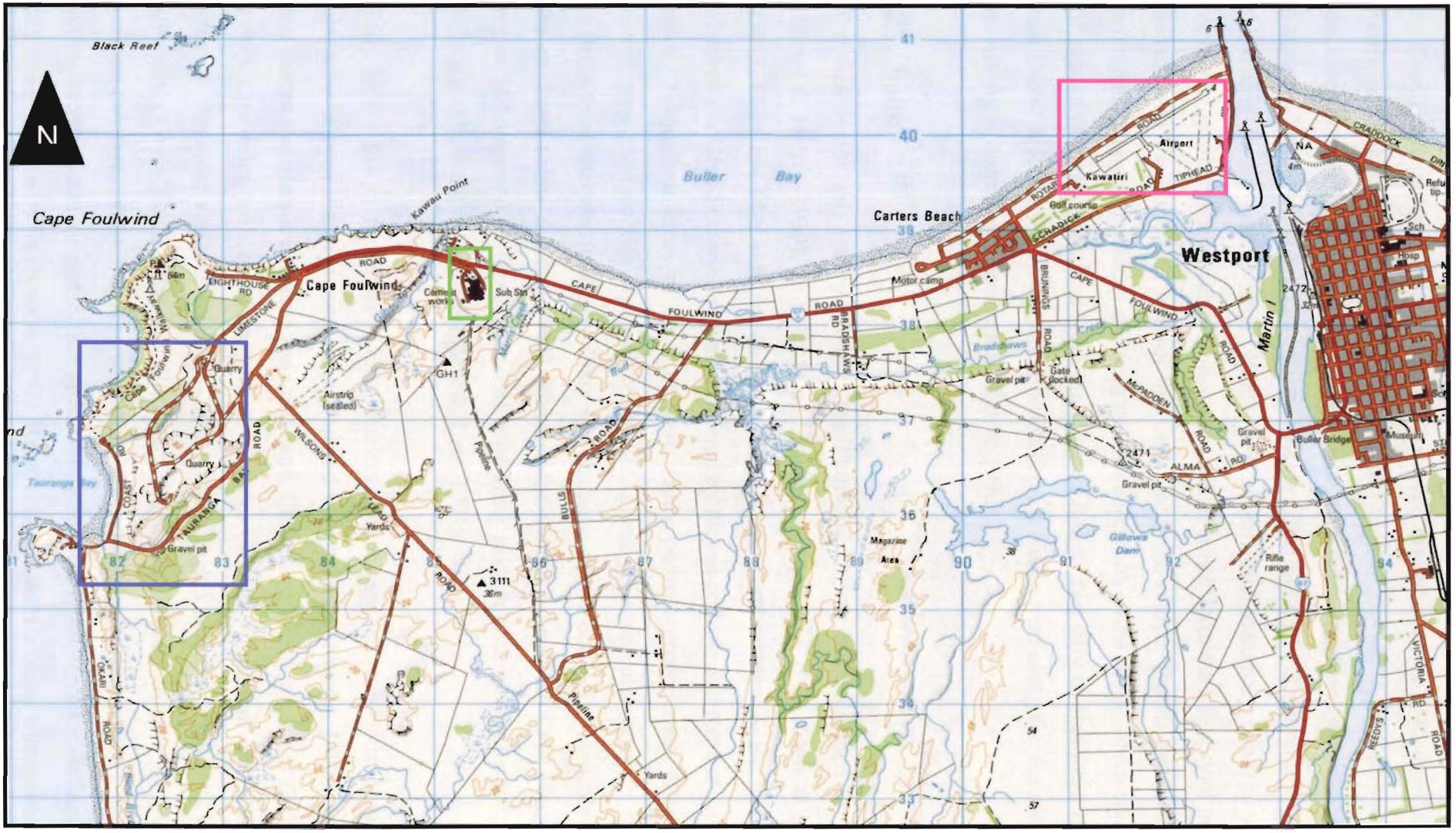


Figure 2.2: Location of the study area (indicated by blue outline). The pink box indicates the location of Westport Airport. The cement works for processing the raw materials from the quarry is outlined in green. Map taken from Dept. of Survey and Land Information, Infomap 242-3 North meets South.

At Cape Foulwind the Tertiary sequence has been draped over the eroded surface of Foulwind Granite and consists of a thin layer of Brunner Coal Measures followed by a thick sequence of the Kaiata Formation, which includes the limestone lens being quarried. This in turn is overlain by the O'Keefe (Blue Bottom) Formation in a wedge thickening to the east (Norton, 1992). The superficial Quaternary deposits are a variety of granitic marine sands with some gravels, which form the majority of parent materials in the north and west of the Cape Foulwind region. Parent materials are principally stony or bouldery alluvium and / or colluvium in the south, which consists of granite and gneiss derived from the Paparoa Range, and silty or fine loess in the east (Mew & Ross, 1991). A coastal beach deposit has formed along the sea margin, consisting of dunes and beach sand (Norton, 1992).

2.3. Soils

The first major soil survey of the Cape Foulwind region was published in 1939 (Harris & Harris, 1939); an updated soil survey took place in 1968 (Mew & Ross, 1991). The Cape Foulwind region has a complex soil pattern (Mew & Ross, 1991). Much of the Cape Foulwind region is covered by gleyed podzols and gleys of various kinds. Throughout areas in the northwest, sandy podzols predominate. Gley and humic gley soils overlie lagoonal sediments. These soils partially separate sandy podzols from the predominantly stony and strongly gleyed podzols, which occur in the southeast (Mew & Ross, 1991). Much of the eastern edge of the Cape Foulwind region is comprised of loessic material overlaying gravels with gley soils and both humus and humus-iron podzols (Mew & Ross, 1991).

Soils on the flat or on gently sloping marine deposits are leached, podzolised and gleyed with poor drainage, and are considered very infertile (Norton, 1992). Where soil parent materials are marine and dune sand, Charleston soils are present (Molloy, 1993). Charleston soils occur on younger surfaces (Waites Formation) and Addison soils on the older surfaces (Norton, 1992), where higher terraces consist of stony alluvium (Virgin Flat Formation). Both surfaces are moderately to strongly gleyed podzols (Molloy, 1993). Surfaces with steep slopes have better drained soils, such as yellow-brown earths, or podzolised yellow-brown earths. However, such surfaces are still likely to be infertile (Norton, 1992). Relatively fertile soils occur where outcrops

of Tertiary parent materials occur, as in the quarry area. Naturally fertile, well-drained soils also occur on coastal dunes and on river flats (Norton, 1992).

2.4. Climate

Climate information has been obtained from two primary sources: Hessell (1982) and the National Institute of Water and Atmospheric Research (NIWA) website (2002). Hessell (1982) data is based on the period from 1944-1978, while the NIWA data is for a more recent thirty year period, 1971-2000. Both sources use data recorded at Westport Airport. This climatological station (formerly known as the Westport Aerodrome) is the only one in the general area. Westport Airport is located approximately 13 km from the quarry at Cape Foulwind (see Figure 2.2) at 41° 44.2′ S latitude and a longitude of 171° 34.5′ E, so slight differences between climatic conditions found at the study area and those reported are possible. However Westport Airport, like the study area, is close to the coast and at low altitude (2 m a.s.l.), making potential differences minimal.

The climate of the Cape Foulwind area can be described generally as mild and humid (Norton, 1992). For the period 1971-2000, the average rainfall was 2 274 mm per year, falling on 169 days (NIWA, 2002). Hessell (1982) reported an average annual rainfall of 2 157 mm for the period 1944-1978. The majority of this rainfall occurs during spring (October, November, and December) and autumn (April and May), each period resulting in 26% of the annual value. Dry spells, where no rain or $\leq$ 1.0 mm falls for a period of at least 15 days, are rare. Despite the high rainfall, the region experiences a relatively high number of sunshine hours; 46% of possible sunshine hours, sharing the same value as Christchurch (Hessell, 1982).

Mean annual temperature recorded in Hessell (1982) is 12.1 °C, with an annual daily range of 7.3 °C. For the more recent period (1971-2000) the average temperature is slightly higher at 12.8 °C, with a maximum temperature of 28.6 °C and a minimum of -3.5 °C. The coldest month is July with an average temperature of 8.6 °C and the warmest month is February with an average of 16.6 °C (NIWA, 2002).

Winds form a prominent feature of the climate at Cape Foulwind. 55% of winds lie between 6 km/h and 30 km/h, and there is a mean wind speed of 11 km/h (NIWA, 2002). The strongest winds in this region occur from the southwest. 43.6% of strong (>30 km/h) winds are from the southwest. Much of the quarry area is particularly exposed to these winds, because of its southwesterly aspect (Figure 2.2; Norton, 1992).

Frosts are rare, with ground frosts occurring on an average of just 27.2 days per year, and screen (1.2 m) frosts even more uncommon, occurring on only 1.0 days a year (Hessell, 1982). Thunder or hail is infrequent in this area, occurring on an average of just 13.7 and 5.8 days annually, respectively. Frequencies of these two phenomena are fairly consistent throughout the year, apart from a slight drop during summer months (Hessell, 1982).

2.5. Human History

The history detailed in the following section is largely adapted from MacDonald (1973). The West Coast of the South Island is recorded as being first seen by Chief Ngahue, during his voyage of circa 950 AD. After this sighting, and prior to the first sighting by a European, which did not occur until 1642, the Ngati Wairangi settled on the West Coast. People settled in coastal environments, which enabled them to obtain their living primarily from the sea. Ngati Wairangi were the initial explorers of this region, and became prosperous by trading in pounamu (or greenstone) with other tribes.

Abel Tasman was the first recorded European to describe and map this part of New Zealand, particularly Cape Foulwind, which he named Clijppijgen Hoeck, in December of 1642. In March 1770, Lieutenant James Cook captained the Endeavour into these same waters in the same direction as previously navigated by Tasman. Unfavourable weather conditions prevented Cook from observing the coastline to any great extent. This is emphasised by the detail that while trying to approach Clijppijgen Hoeck, the Endeavour was blown several miles off course to the southwest. On regaining the desired location, Captain Cook appropriately renamed Clijppijgen Hoeck, Cape Foulwind.

Sealers were the earliest recorded European settlers of the West Coast, scattered up and down the coast from as early as 1792, establishing base camps ashore while searching for and collecting valuable oil and furs. One such base was situated at Black Reef, near Cape Foulwind.

Commander Jules Sébastien César Dumont d'Urville, sighted the Cape Foulwind region in 1827, commenting that no inhabitants could be seen, but if there were any "...they must have settled in the neighbourhood of Cape Foul Wind, where the telescope revealed some attractive sites, and fine grass-land suitable for cultivation." D'Urville had been deceived by the open appearance of the poorly drained and infertile pakihi immediately behind Cape Foulwind.

The township of Westport was formed as the result of the discovery of "payable" gold in the Buller River (MacDonald, 1973). In 1884 the Westport Harbour Board was formed. Shortly thereafter, the Harbour Board began to acquire stone. After bore testing at Te Kuha and Cape Foulwind it was decided to open quarries at Cape Foulwind, which was done so in 1886. Blasting began on 15 January 1886.

Timber was in high demand during the early period of European settlement as it was a necessity for mining and railway construction. A major sawmill operated near Cape Foulwind during the period from the 1870s through to 1920s, utilising both beech for coalmines and silver pine, used extensively for railway sleepers (MacDonald, 1973; Norton, 1992).

Gold mining occurred on the Cape Foulwind flats, both on the higher terraces around SH 6 and in the upper reaches of Bradshaw Creek, particularly during the 1860s and 1870s. Gold mining would have been associated with much forest clearance, probably including the use of fire (Norton, 1992; MacDonald, 1973).

2.6. Vegetation History

Limited historical accounts of vegetation were available; therefore the following description has been adapted from Norton (1992). The general area of this study was apparently first botanised by Townson (1907), however a specific description of

vegetation that occurred on the flats at Cape Foulwind does not appear to have been published. Norton (1992) based his description of the vegetation history of the area on remnants that remain on the flats in addition to vegetation patterns elsewhere in north Westland and Buller, as little original vegetation remains.

Topography, soils and proximity to the coast are the three factors that have played the greatest influential role on the vegetation patterns present in the Cape Foulwind flats (Norton, 1992). A mixed beech-podocarp forest, due to shallow infertile and wet podzol soils, would have dominated the largest area of the flats; the relative dominance of each component dependent on soil moisture. At elevated sites *Nothofagus truncata* would have been the dominant canopy tree, with scattered emergent *Dacrydium cupressinum* and *Prumnopitys ferruginea* (Norton, 1992). Podocarps would have become more prevalent, and beech less abundant or absent in low-lying areas. Podocarp species, in addition to *D. cupressinum* and *P. ferruginea*, would have included *Manoao colensoi*, *Phyllocladus* sp. aff. *alpinus* and *Metrosideros umbellata.* Other associated small tree and shrub species likely to have been present would have included *Weinmannia racemosa*, *Quintinia serrata*, *Myrsine salicina*, *Lophomyrtus obcordata*, *Elaeocarpus hookerianus* and *Coprosma* species. *Nothofagus* spp. do not appear to have been present north of Blind Valley, including the quarry area.

Near the coast, especially on young sediments (Nine-Mile Formation) and Tertiary deposits, the forest composition appears to have been quite different. *Dacrycarpus dacrydioides* would have been common on recent fertile soils, close to the coast especially on young sediments (Nine-Mile Formation) and Tertiary deposits. Tertiary deposits near Cape Foulwind appear to have hosted a much more diverse forest including canopy species such as *Melicytus ramiflorus*, *Macropiper excelsum*, *Myrsine salicina*, *Aristotelia serrata*, *Elaeocarpus dentatus*, and *Coprosma grandifolia*, possibly with *Dacrydium cupressinum*, *Dacrycarpus dacrydioides*, and *Metrosideros robusta* emerging over this canopy. Vines would have been abundant, particularly *Ripogonum scandens* and *Freycinetia baueriana*. *Rhopalostylis sapida* would have occurred in sheltered coastal locations, with *Cordyline australis* common around active streams. Small tree and shrub species (*Melicytus ramiflorus*,

Macropiper excelsum and *Phormium tenax*) would have dominated windswept coastal sites (Norton, 1992).

2.6.1. Vegetation Modification

It is apparent that forests were modified substantially during the early periods of human settlement for the purpose of providing timber for coalmine and railway construction. Gold mining was also associated with much forest clearance (Norton, 1992; MacDonald, 1973).

Today, the vegetation of the Cape Foulwind area is dominated by pasture and pakihi, including a few small forest remnants. Much of the farmland is only partially developed with large amounts of rush present (Norton, 1992). Most of the remaining forest is Crown land managed under a Department of Conservation covenant (especially on the LandCorp Cape Foulwind farm). Small forest remnants occur on land owned by Holcim.

2.7. Study Site Selection

Nine sites located around the Holcim quarry were used to assess the initial success of the company's ecological restoration efforts. Six study sites were located in planted areas of various ages (time since planting) and three study sites occurred in naturally regenerating forest located on Holcim land. The three remnant study sites lay in close proximity to the plantings and were used as comparisons with the planted restoration study sites for the purpose of this research (Figure 2.3).

2.7.1. Description of Study Sites

This section provides a brief description of the vegetation and elaborates on other characteristics of each site.

Planted Restoration Site 1 (P1)

25,000 plants were planted on this site in the months following June 1999. The P1 study site is situated on a gently sloping hill (7.4° ± 1.4) comprised of aged and weathered quarry strippings. Study plots established in this area predominately faced

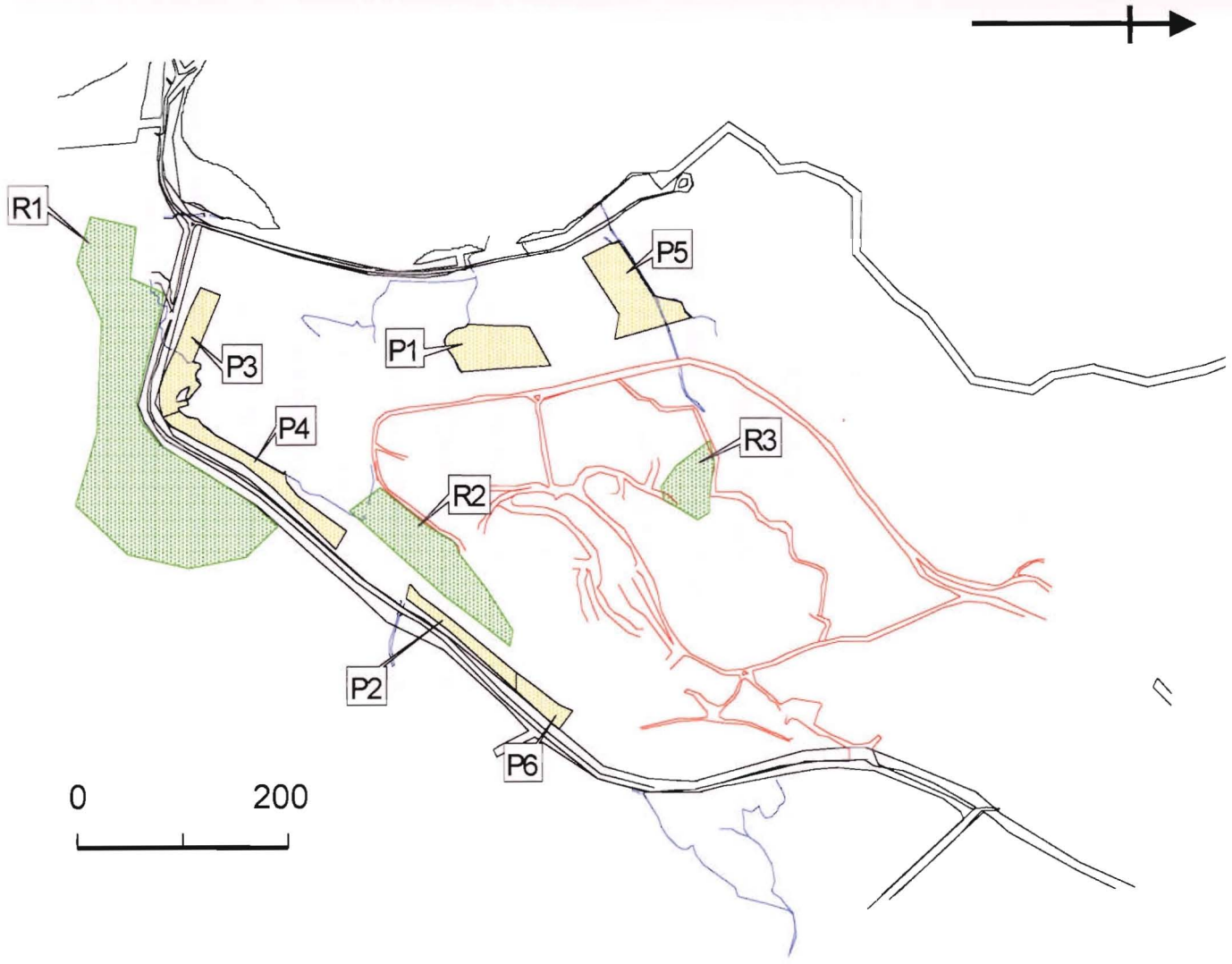


Figure 2.3: Location of study sites within the quarry area. Areas blocked in green represent remnant study sites, while tan coloured areas represent the six planted restoration study sites.

in a southwesterly direction and accordingly, much of the area is directly exposed to strong onshore, salt laden winds.

Preparation of the site, prior to planting, involved eradication of *Ulex europeaus,* then blanket spraying for the established pasture grass cover that was present. Common tree and shrub species planted in this study area include *Hebe elliptica*, *Phormium tenax*, *Leptospermum scoparium*, with numerous *Coprosma robusta* and *Myoporum laetum* present also. Post-planting maintenance of this area has involved release spraying using a combination of Versatil® and Galant®; grass specific herbicides. Each plant also received a hand application of a combination of quick- and slow-release fertilisers (Keir, 1999; 1998).

Planted Restoration Site 2 (P2)

P2 was formed from ironsand quarry strippings, and is located alongside Tauranga Bay Road. Only the roadside of the berm was investigated during this study. The berm has an average slope of 19.2° ± 0.9, and is positioned in an east to southeast aspect. Ironsand has a high erosion factor; therefore the whole site was sown with a mixture of ryegrass and clover seed in 1997 prior to planting, to reduce the risk of erosion. Topsoil was also spread over the site. Approximately 17 000 plants were planted during the autumn and winter period of 1997. Release spraying using the combination of grass specific herbicides, Versatil® and Galant® has been an integral part of post-planting maintenance, in addition to multiple applications of top dressing fertiliser since planting. The main factors affecting this study site are the poor quality soil and its inability to supply moisture in dry weather (Keir, 1998), and extensive growth of exotic grass and herb species, stifling the growth of planted trees and shrubs. Principle species planted in this area include *Coprosma robusta*, *Hebe elliptica*, *Coprosma propinqua*, and *Leptospermum scoparium.*

Planted Restoration Site 3 (P3)

The planting of this area was undertaken during spring of 1996. The area consists of low rolling sand dune hills, which prior to planting, were grazed by cattle for many years. Two distinct microclimatic zones occur over this study site. A portion of P3 is directly exposed to the salt laden winds from the southwest, facing Tauranga Bay and includes plantings opposite the Tauranga Bay nursery gate. A more sheltered area lies

along Tauranga Bay Road and includes a low-lying gully behind the aforementioned hills. Although sheltered from maritime winds, alternative issues of wetness and poor drainage plague the more sheltered areas (Keir, 1998). Much of the area faces south, with an average slope of 10.7° ± 1.9.

P3 has a loamy, sandy soil structure, and prior to planting was covered by pasture grasses, weeds, and intermittent *Ulex europeaus*. 20 000 plants were planted over a two month period during spring 1996. *Hebe elliptica* is the dominant species planted. Other common planted species include *Hebe salicifolia*, *Pittosporum colensoi*, *Coprosma propinqua*, and *Coprosma robusta*. Difficulties encountered during plant establishment resulted form the grass sward and continuing growth of *Ulex europeaus*, both of which require post-planting management through the use of herbicides, and chainsaw. Applications of quick and slow-release fertilisers have occurred multiple times since planting (Keir, 1998).

Planted Restoration Site 4 (P4)

This study site is situated along Tauranga Bay Road between study sites P2 and P3. P4 has a primarily westerly aspect, with a mean slope of 6.1° ± 1.3. Plantings were done initially by Newton-White in 1996. A lack of maintenance and subsequent *Ulex europeaus* growth resulted in the loss of numerous plants. During the winter and summer months of 1997, gorse was released from the study area by chainsaw and herbicide application. Maintenance spraying of grass was not deemed necessary, as it was determined that plants which had survived up to that point were at such a height as not to be affected by grasses and weeds (Keir, 1998). Infill planting took place in winter 1999. The area is taken up by clumped plantings with large open areas of exotic grass cover between. Dominant planted species present in P4 include *Coprosma robusta*, *Phormium tenax*, *Pittosporum colensoi*, *Coprosma propinqua*, and *Cordyline australis*.

Planted Restoration Site 5 (P5)

This study site borders Department of Conservation land at Cape Foulwind, directly opposite Tauranga Bay, and has a southwesterly aspect. Newton-White undertook the plantings in 1996. The study site is a sand flat that was a grazed paddock prior to planting. The area has a gentle slope of just 5.2° ± 0.8. P5 is occupied by clumped

plantings of primarily *Phormium tenax*, *Coprosma robusta* and *Hebe elliptica* with large open areas of exotic grass and rushes, due to different levels of drainage. There was initially high plant mortality resulting from a decision to use weed mats rather than herbicide to control the grass sward. Unfortunately the weed mats were not successful and many plants were smothered by the invasive grass.

Planted Restoration Site 6 (P6)

This study site was planted in 1980 and is located along Tauranga Bay Road, near the entrance to the quarry. The area has a steep slope of 28.1° ± 0.8 and a northerly aspect. *Leptospermum scoparium*, *Cortaderia richardii*, *Pittosporum tenuifolium* and *Cordyline australis* are the dominant planted woody species in this study site. Naturally established fern species are also abundant; particularly *Pteridium esculentum*. The boundaries of P6 include only areas planted in native species. Large areas along this berm have been planted using exotic species, and as such were deemed to be part of a 'beautification' process as opposed to ecological restoration, hence were not included in the bounds of the P6 study area.

Remnant Site 1 (R1)

R1 is located behind the Tauranga Bay nursery, with an easterly aspect. R1 has many rolling hills, resulting in an average slope for the study area of 12.1° ± 1.7. *Weinmannia racemosa* is a particularly common species in this remnant, with numerous individuals of *Melicytus ramiflorus*, *Myrsine salicina*, *and Dacrycarpus dacrydioides*, present as canopy species at approximately 10 m in height. *Coprosma grandifolia* is an abundant shrub species, present up to 4 m in height. Lianes, particularly *Freycinetia baueriana* and *Ripogonum scandens*, and tree ferns are common in the forest interior. Abundant regenerating tree seedling species were present, particularly *Hedycarya arborea*, *Coprosma grandifolia*, and *Dacrycarpus dacrydioides*. Seedlings of *Rhopalostylis sapida*, *Weinmannia racemosa* and *Ripogonum scandens* were all locally common. A dense litter layer was also present, with native grasses, *Uncinia* spp. common.

Remnant Site 2 (R2)

In R2, the forest canopy, typically 6 m in height, is dominated by *Myrsine salicina* and *Weinmannia racemosa*. Other common canopy species include *Dacrycarpus*

dacrydioides, *Prumnopitys ferruginea*, *Melicytus ramiflorus*, and *Dacrydium cupressinum*. *Coprosma grandifolia* and *Dicksonia squarrosa* are common in the shrub stratum (< 4 m). Regeneration of woody species in the forest interior is common; particularly *Dacrycarpus dacrydioides*. Seedlings of *Coprosma grandifolia* also occurred frequently, with some *Hedycarya arborea* and *Parsonsia heterophylla*. Ground cover was predominantly a dense litter layer. R2 has slight slope of just 5.2° ± 0.8, with an aspect of a southwesterly direction.

Remnant Site 3 (R3)

R3 occurs within the bounds of the quarry proper, facing in a southwesterly direction. The average slope for the study area is 11.9° ± 1.5. The low forest present in R3 consists of a canopy, largely 4-6 m tall, dominated largely by *Coprosma grandifolia* with frequent occurrences of *Hedycarya arborea* and *Melicytus ramiflorus*. *Dicksonia squarrosa* is also common. A dense re-growth of subcanopy and canopy species often occurs on this forest floor, with *Rhopalostylis sapida* seedlings locally common. *Macropiper excelsum* is a distinct forest interior species, occurring as both a shrub and regenerating seedlings. A dense litter layer dominated the ground cover in R3.

3. VEGETATION COMPOSITION

3.1. Introduction

The overall objective of this study was to determine the success of the restoration plantings to date. Vegetation provides an immediate visual indication of the state of restoration plantings via the assessment of species composition, through to the detection of ecosystem processes such as regeneration, which is indicative of the occurrence of dispersal (an ecosystem process).

The purpose of this chapter is to: (1) provide an insight into the diversity of vegetation in the nine study sites, and (2) investigate the ability of the restoration plantings to facilitate regeneration. More specifically this chapter seeks to test whether:

1. species composition varied between planted restoration sites and remnants;
2. species composition varied between the six planted restoration sites;
3. species and abundance of regenerating seedlings of woody species are dependent on the dominance of canopy species
4. restoration plantings are facilitating regeneration; and whether
5. growth of woody species within planted restoration sites differs with time since planting.

3.2. Methods

3.2.1. Field Methods

Vegetation composition was investigated using twenty 4 × 4 m plots, arranged in stratified random design, within each of the nine study sites. This stratified random design involved a series of four transects of unequal length, laid out within each study site such that all parts of the study sites would be sampled (Figure 3.1). Transects were set out at unequal distances from each other. Each transect had five study plots situated along it, alternating from the left to right side of each transect, at equal distances from each other. All distances used to establish study plots were randomly determined.

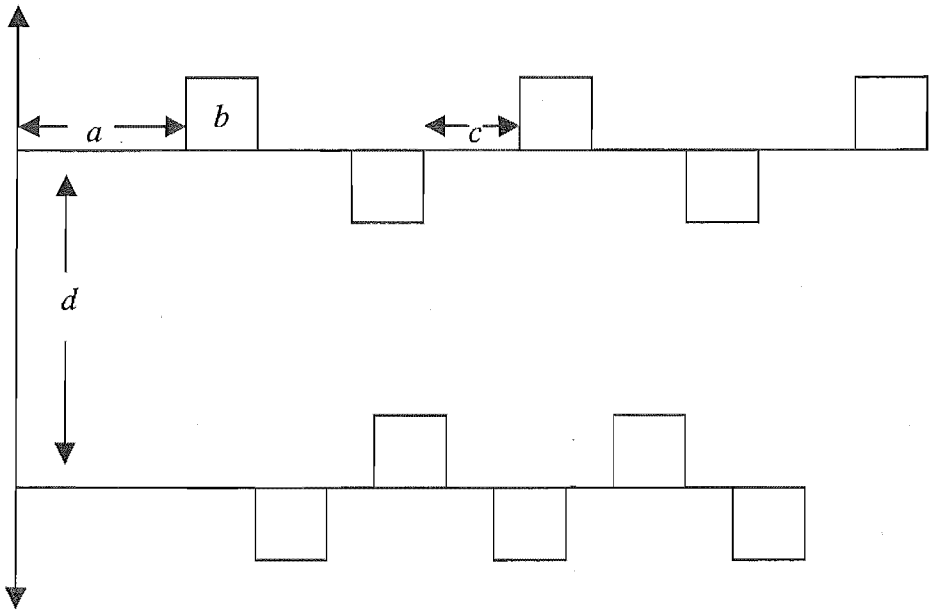


Figure 3.1: Example of arrangement of study plots, in stratified random design, within each of the nine study sites. *a* represents the distance from the start line of the transect through to the first study plot. *b* represents the 4 × 4 m study plots used to estimate vegetation composition. *c* represents the distance between study plots along transect lines. *d* shows the random distance in between transect lines.

Within all study plots, vertical and horizontal cover estimates were completed for each species. Kent & Coker (1992) describe cover as the area of ground within a quadrat occupied by the above ground parts of each species, when viewed from above. Cover is one of the most widely used measures of abundance for plants species as it is not biased by size or distribution of individual plants (Floyd & Anderson, 1987) and is a rapid method of assessment to use (Kent & Coker, 1992). Vertical strata were divided into appropriate height divisions, namely < 0.3 m, ≥ 0.3-2 m, ≥ 2-5 m, and ≥ 5-12 m, based on vegetation structure within individual plots. Cover estimates were completed for each species, as well as for all species together in each stratum. Horizontal cover within plots was estimated using seven cover abundance classes: < 1%, 1-5%, 6-10%, 11-25%, 26-50%, 51-75%, and 76-100%. Cover estimates were made from the centre of each plot for consistency.

As cover estimates were done by eye, a certain degree of human error was inevitably involved with data collection. Possible sources of error included incorrect identification of species, failure to consistently measure all plants in a sample plot in the same way, and difficulty in determining the vertical boundaries of some plants. Kent & Coker (1992) note that species that are conspicuous, attractive or in flower

tend to be overestimated, while species unknown to the recorder tend to be underestimated. As all identification and estimates of vegetation cover were undertaken by the same observer, measurements should be comparable across each of the nine study sites.

Density of regenerating woody seedlings was recorded and classed into two height categories: 0.02-0.1 m, and ≥ 0.1-0.3 m. Groundcover assessments were completed using cover classes detailed previously. Groundcover was classified as native woody, exotic woody, native herbs, exotic herbs, native grasses, exotic grasses, rushes, ferns, bryophytes, litter, exposed soil, or exposed rock.

3.2.2. Data Analysis

In order to clarify analyses, vegetation data were grouped into three classes:

1. regeneration (of woody species),
2. ferns and allied plants, and
3. all woody species (including climbers).

Data analysis was completed using appropriate data only. For example, diversity assessments for vegetation in the 'all woody species' category, all appropriate species were included and others, such as fern species, excluded from analysis.

3.2.3. Importance Values

Cover estimate values of vegetation were modified to obtain a single importance value for each species in each plot. Importance values were calculated by multiplying the log10 of each stratum height (+1), by the midpoint of the species cover class (Hall, 1992). Where applicable (within remnant study site plots), importance values for shrub, sub-canopy and canopy strata were combined to provide a single importance value for a 'canopy' stratum. Importance values were used for all analyses, with the exception of species richness and floristic similarity coefficients, where presence-absence data was relevant.

3.2.3.1. Diversity Indices

Diversity indices are used frequently in ecological studies as an indication of the wellbeing of ecosystems (Magurran, 1988). Diversity is measured by recording the number of species and their relative abundances within a sample (Kent & Coker,

1992). Six diversity indices were used in this study to enable comparisons between the nine study sites. Values for each diversity index were calculated based on mean values across the twenty study plots within each study site. For each diversity index, following calculation, statistical tests were run to determine whether differences were present between mean values for the nine study sites (see Section 3.2.3.2.).

Species Richness

Species richness (S), the number of species within a chosen site, is the definition of alpha (α) diversity, and is the most common diversity measure used in ecological studies (Magurran, 1988). Species richness was recorded as the total number of species observed within each study site, for each of the three vegetation categories.

Cover

This analysis was undertaken using total importance values for each species. Where appropriate (i.e. in remnant sites) importance values were combined for each species in shrub, sub-canopy, and canopy levels. Cover is a subjective measure of species abundance (Kent & Coker, 1992).

Heterogeneity

Shannon's diversity index (H') is a measure of heterogeneity, combining species richness and evenness into a single measure (Lapin & Barnes, 1995). The Shannon index is an example of a species abundance model, which is the most complete mathematical description of the data as it uses all information gathered in a community (Magurran, 1988). Shannon and Wiener arrived independently at this diversity index, which has since become known as the Shannon index (Magurran, 1988). The Shannon index is:

$$H' = -\sum_{i=1}^{s} p_i \ln p_i$$

where p_i is the proportion of abundance represented by the ith species. In a sample, such as that used in this study the actual value of p_i is unknown, but is estimated as n_i / N (the maximum likelihood indicator), where n_i = the amount of cover of the ith species, and N = the total amount of cover in the sample (Kent & Coker, 1992). Use of n_i / N, as an estimator of p_i produces a biased result; however Magurran (1988) notes this bias to rarely be of significance.

There are two primary assumptions to the Shannon index: (1) that individuals are randomly sampled from an infinitely large population, and (2) that all species present in a population are represented in the sample. The latter assumption is often a significant source of error, which increases as the proportion of species represented in a sample decreases (Magurran, 1988).

Magurran (1988) states that *H'* values commonly fall between 1.5 and 3.5, and rarely surpass 4.5.

Dominance

The Berger-Parker index is a dominance measure, meaning it is weighted towards the abundances of the commonest species rather than providing a measure of species richness. Thus, it is an expression of the proportional importance of species (Magurran, 1988). The Berger-Parker index is:

$$d = \frac{N_{\max}}{N}$$

where N_{max} = the amount of cover of the most abundant species and N = the total cover of individual species in the sample. The reciprocal form of this index was adopted in this study, so that an increase in the value of this dominance index accompanies an increase in diversity and a reduction in dominance (Magurran, 1988).

Evenness

Evenness is a measure of how evenly species abundances are distributed within a community (Alatalo, 1981). The modified Hill's ratio was adopted for use in this study:

$$E = \frac{\left(1/\sum p_i^2\right)-1}{\left(\exp\left(-\sum p_i \ln p_i\right)\right)-1}$$

where p_i is the proportion of cover represented by the ith species.

Floristic Similarity

Floristic similarity is a simple measure of beta (β) diversity, i.e. how similar or different study sites are. A widely used similarity coefficient is Jaccard's coefficient:

$$C_J = \frac{j}{(a+b-j)}$$

where *j* equals the number of species present in both samples, *a* represents the number of species present in site 1, and *b* equals the number of species present in site 2 (Kent & Coker, 1992). This index equals one when sites exhibit complete similarity, i.e., where the two sets of species are identical, and zero if the sites are completely dissimilar.

A disadvantage of this method is that Jaccard's coefficient utilises only presence-absence data, not taking species abundance into account (Magurran, 1988). Presence-absence was first calculated between plots within each study site and then compared across study sites.

3.2.3.2. Statistical Analysis

Diversity assessments for the nine study sites, except floristic similarity, were compared statistically using one-way analysis of variance (ANOVA), with the aid of the statistical package SAS V8. ANOVA was used to compare between planted restoration and remnant study sites, as well as between the six planted restoration sites. Pairwise multiple comparisons were then run using the Tukey's test to determine the nature of differences detected by ANOVA. The level of significance set for all statistical testing was set at $\alpha = 0.05$.

3.2.3.3. Mode of Dispersal

In order to assess whether or not restoration plantings were facilitating dispersal, dispersal mode of all seedlings recorded was categorised as bird or wind for each study site. This classification was dependent on whether seeds were available to birds as a food source in the form of a berry or drupe (Reay, 1996; Burrows, 1994).

3.3. Results

3.3.1. Regeneration

3.3.1.1. Diversity Assessments

Species Richness

An ANOVA run for species richness of regenerating seedlings present by type of study site (planted restoration or remnant sites) indicated that a significant difference was present ($F = 218.30$, df = 1, $P = < 0.001$). According to the Tukey's test, remnant sites had a significantly higher mean number of different species of regenerating seedlings present than planted restoration sites. Similarly, a significant difference in mean species richness of regenerating seedlings was suggested in planted restoration sites ($F = 4.61$, df = 5, $P = < 0.001$). The Tukey's test revealed that P6 had significantly more species of regenerating seedlings, than P2 and P1 (Table 3.1).

Table 3.1: Species richness of regenerating seedlings of woody species. Part (a) of the table investigates the difference in the number of species of regenerating seedlings present in remnant and planted restoration sites, while part (b) looks at descriptive statistics for species of regenerating seedlings in the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.0	7.0	4.3^{A}	0.2
	P	120	0.0	6.0	0.8^{B}	0.1
(b)	P1	20	0.0	0.0	0.0^{B}	0.0
	P2	20	0.0	1.0	0.1^{B}	0.1
	P3	20	0.0	6.0	0.9^{AB}	0.4
	P4	20	0.0	5.0	1.1^{AB}	0.4
	P5	20	0.0	6.0	0.8^{AB}	0.3
	P6	20	0.0	5.0	1.8^{A}	0.4

R2 and R3 had the largest total number of species of regenerating seedlings of woody species present with $S = 13$. P1 lacked evidence of any regeneration, while P3, P5 and P6 shared the highest total number of regenerating seedling species ($S = 8$) for the whole site, in planted restoration sites (Figure 3.2a).

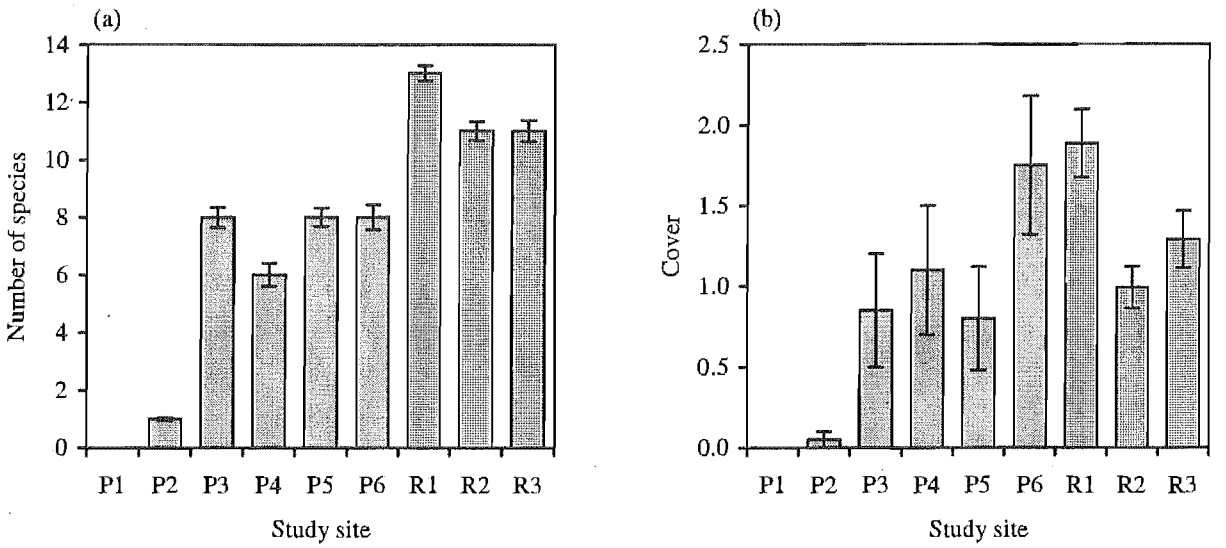


Figure 3.2: Graph (a) illustrates the relationship between study sites in terms of species richness (*S*) of regenerating seedlings, present in two vertical strata (0.02-0.1 m, and >0.1-0.3 m); graph (b) shows the cover of seedlings in each study site. Error bars are the standard error around the mean.

Cover

Planted restoration sites varied significantly from remnant sites in terms of cover occupied by regenerating seedlings ($F = 22.84$, df = 1 $P = < 0.001$). According to the Tukey's test, significantly less cover was represented by regenerating seedlings in planted restoration sites than in remnants. The results of an ANOVA for cover by planted restoration sites suggested that significant differences were present in the six study sites ($F = 7.86$, df = 5 $P = < 0.001$). Tukey's test elaborated on this suggested difference by revealing that P6 had a significantly greater cover of regenerating seedlings than did all other planted restoration study sites (Table 3.2). R1 had the largest cover values for regenerating seedlings (1.9) for all study sites; while P6 had the greatest cover value for regenerating seedlings in planted restoration sites (1.8). P1 was without a cover value, due to the lack of evidence of regeneration (Figure 3.2b).

Table 3.2: Descriptive statistics values for cover represented by regenerating seedlings of woody species. Part (a) of the table looks at descriptive statistics for remnants and planted restoration sites, while part (b) compares cover values of the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.0	4.1	1.4^{A}	0.1
	P	120	0.0	9.1	0.5^{B}	0.1
(b)	P1	20	0.0	0.0	0.0^{B}	0.0
	P2	20	0.0	0.2	0.0^{B}	0.0
	P3	20	0.0	1.4	0.2^{B}	0.8
	P4	20	0.0	1.2	0.3^{B}	0.1
	P5	20	0.0	2.9	0.4^{B}	0.2
	P6	20	0.0	9.1	2.0^{A}	0.6

Heterogeneity

According to ANOVA, remnant sites differed significantly from planted restoration sites in terms of heterogeneity (F = 250.67, df = 1, P = < 0.001). It was shown by Tukey's 'comparison of means' test that planted restoration sites were significantly less heterogeneous than remnant sites. ANOVA detected a significant difference between H' in planted restoration sites (F = 3.34, df = 5, P = 0.008). Tukey's test revealed that P6 had significantly higher values for H' than P1 and P2, which both had zero values (Table 3.2). R1 had the greatest value for the heterogeneity index (H' = 1.51). P1 and P5 had zero values. P6 was the most heterogeneous of the planted restoration study sites (H' = 0.40) (Figure 3.3a). The H' value for R1 alone sat within the boundary of which most ecological data sets are said to lie (H' = 1.5-3.5) (Magurran, 1988).

Table 3.3: Descriptive statistics values for the Shannon index (H') for regenerating seedlings. Part (a) of the table investigates the difference in H' values in remnants and planted restoration sites. Part (b) compares H' values for all six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.00	1.95	1.29^{A}	0.06
	P	120	0.00	1.79	0.20^{B}	0.04
(b)	P1	20	0.00	0.00	0.00^{B}	0.00
	P2	20	0.00	0.00	0.00^{B}	0.00
	P3	20	0.00	1.79	0.23^{AB}	0.11
	P4	20	0.00	1.61	0.35^{AB}	0.13
	P5	20	0.00	1.79	0.18^{AB}	0.10
	P6	20	0.00	1.27	0.40^{A}	0.11

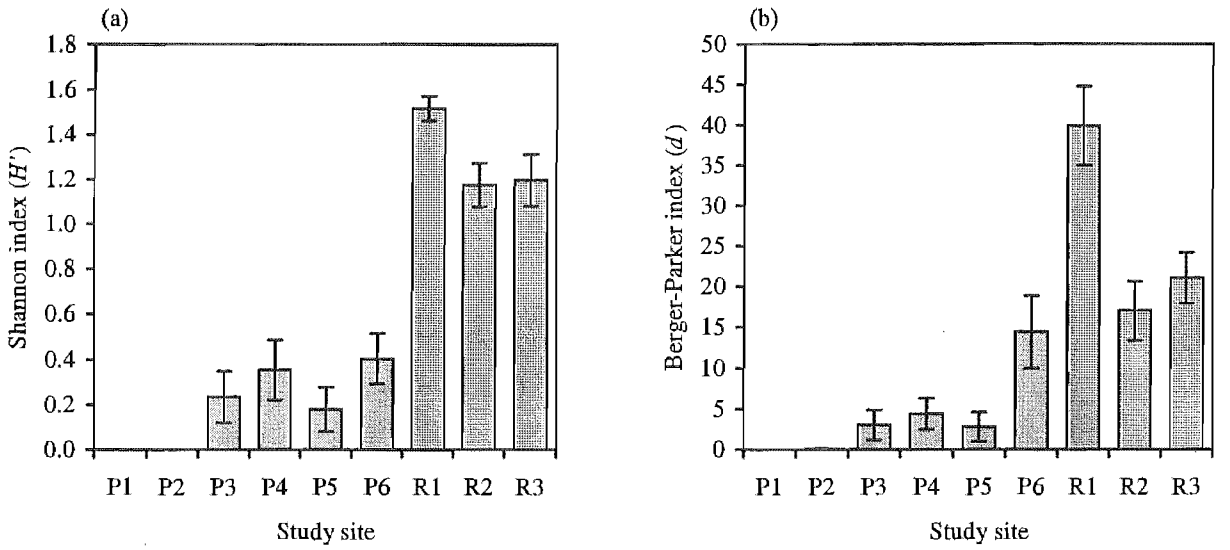


Figure 3.3: Graphs (a) and (b) illustrate the relationship of regenerating seedlings between study sites using two heterogeneity indices: Shannon's diversity index and the reciprocal of the Berger-Parker index, respectively. The error bars depict the standard error around the mean.

Dominance

P6 had the greatest value for the reciprocal of the Berger-Parker index (d = 14.42), indicating comparatively less dominance of one species of regenerating seedlings, than P2 (d = 0.05), for which only one seedling was detected (Figure 3.3b). R1 had the greatest value for the Berger-Parker index suggestive of the diversity of regenerating seedlings found within this study site, and the resulting lack of dominance.

A significant difference was detected for dominance values between planted restoration and remnant sites (F = 90.47, df = 1, P = < 0.001). The Tukey's test showed that remnant sites had significantly greater mean d values than planted restoration sites, which reflect the lesser dominance of a particular, or very few species of seedlings found in remnant sites. A significant difference was also found between the six planted restoration sites (F = 5.78, df = 5, P = < 0.001). The results of the Tukey's test showed that this statistically significant difference was due to P6 which had a greater diversity of regenerating seedlings than the other five planted

restoration sites, thereby obtaining a higher *d* value illustrating the lack of dominance of a singular species (Table 3.4).

Table 3.4: Descriptive statistics for the reciprocal of the Berger-Parker index (*d*) for regenerating seedlings. Part (a) of the table looks at descriptive statistics for remnants and planted restoration sites, while part (b) shows values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.00	90.67	25.98^{A}	2.58
	P	120	0.00	58.19	4.09^{B}	0.99
(b)	P1	20	0.00	0.00	0.00^{B}	0.00
	P2	20	0.00	1.00	0.05^{B}	0.05
	P3	20	0.00	36.00	3.05^{B}	1.84
	P4	20	0.00	25.00	4.42^{B}	1.91
	P5	20	0.00	36.00	2.81^{B}	1.81
	P6	20	0.00	58.19	14.42^{A}	4.43

Evenness

An ANOVA run for the modified Hill's ratio for evenness of regenerating seedlings by type of study site (remnant or planted restoration study sites) suggested that a statistically significant difference existed between the two types of study sites (F = 188.61, df = 1, P = < 0.001). According to the Tukey's comparison of means test, remnant sites had significantly higher average evenness values. Additionally, statistically significant differences were found for evenness of regenerating seedlings between the six planted restoration sites (F = 3.71, df = 5, P = 0.004) (Table 3.5). P6 was found to have a significantly more even distribution of regenerating seedlings than the P1 and P2 study sites. P6 had the highest value (E = 0.34) for the modified Hill's ratio for evenness (Figure 3.4). While similar to other diversity indices for regenerating seedlings, P1 and P5 obtained zero values for evenness. R2 was the most even study site (E = 0.91).

Table 3.5: Descriptive statistics values for the modified Hill's ratio for evenness (E) of regenerating seedlings. Part (a) of the table looks at descriptive statistics of evenness values in remnants and planted restoration sites, while (b) compares evenness values in the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.00	1.13	0.87^{A}	0.03
	P	120	0.00	1.00	0.17^{B}	0.03
(b)	P1	20	0.00	0.00	0.00^{B}	0.00
	P2	20	0.00	0.00	0.00^{B}	0.00
	P3	20	0.00	1.00	0.20^{AB}	0.09
	P4	20	0.00	1.00	0.30^{A}	0.11
	P5	20	0.00	1.00	0.18^{AB}	0.09
	P6	20	0.00	1.00	0.34^{A}	0.08

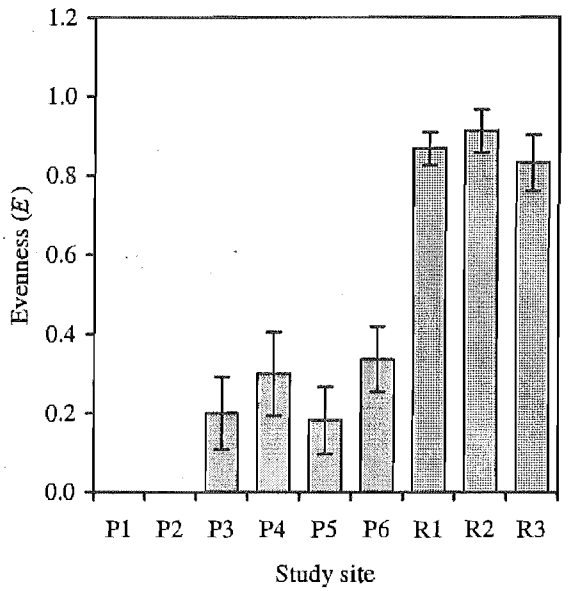


Figure 3.4: Comparison of mean values for the modified Hill's ratio for evenness (E) of regenerating seedlings of woody species for the nine study sites. The standard error about the mean for each study site is illustrated.

Floristic Similarity

Low Jaccard's coefficients were common when comparing species of regenerating seedlings located within the nine study sites. P1 was completely dissimilar to all other study sites, due to the lack of evidence of regeneration within this study site. P2 had very low similarity to all remnant sites (Table 3.6). R1 and R2 were the study sites that shared the most regenerating species in common (C_J = 0.60). Remnant and planted restoration study sites shared almost 50% of species of regenerating seedlings in common (C_J = 0.43).

Table 3.6: Jaccard's similarity coefficients for floristic similarity of regeneration of woody species between study sites.

	P1	P2	P3	P4	P5	P6	R1	R2	R3
P1	0.00								
P2	0.00	1.00							
P3	0.00	0.13	1.00						
P4	0.00	0.17	0.56	1.00					
P5	0.00	0.13	0.23	0.40	1.00				
P6	0.00	0.13	0.45	0.40	0.14	1.00			
R1	0.00	0.08	0.50	0.27	0.11	0.50	1.00		
R2	0.00	0.09	0.46	0.21	0.12	0.46	0.60	1.00	
R3	0.00	0.09	0.27	0.21	0.19	0.27	0.60	0.47	1.00

3.3.1.2. Mode of Dispersal

The principal mode of dispersal for regenerating woody species located in any of the nine study sites was bird dissemination. This dominance of bird dispersed seedlings is evident both in the dispersal mode of different species located in different sites, and the proportion of the total number of seedlings located dispersed by either method (Figure 3.5).

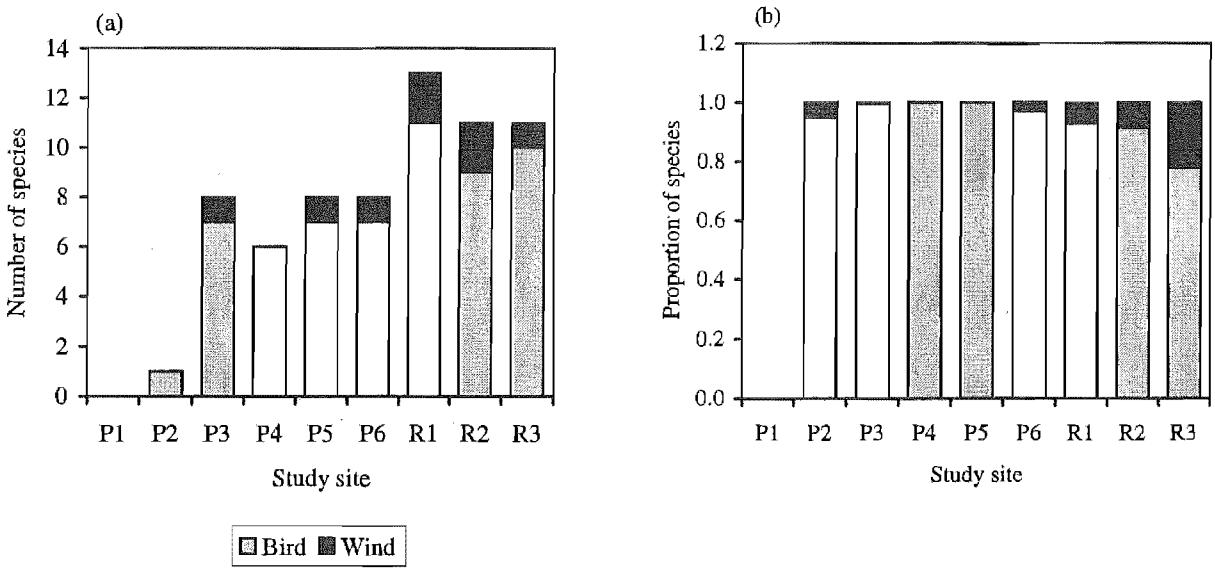


Figure 3.5: Mode of dispersal. Graph (a) illustrates the number of different seedling species found in each study site with a particular dispersal mode. Graph (b) illustrates dispersal mode further by showing the proportion of regenerating seedlings of each particular mode of dispersal. The key, indicating dispersal mode, is applicable to both graphs.

Study sites P3, P5 and P6 all had seven species of bird dispersed seedlings, and only one wind dispersed species detected. P4, P2, and P1 did not have any wind / gravity dispersed species within study plots. Despite three wind dispersed species found as regenerating seedlings within planted restoration sites, very few wind dispersed seedlings were measured present as a proportion of the total number of seedlings found.

Table 3.7: Dispersal mode of regenerating seedlings found throughout planted restoration and remnant sites. The symbol √ indicates the presence of a particular species in either of the two types of study site.

Species	Dispersal Mechanism	Planted Restoration Study Sites	Remnant Study Sites
Aristotelia serrata	bird	√	
Carpodetus serratus	bird	√	√
Coprosma areolata	bird		√
Coprosma grandifolia	bird	√	√
Coprosma propinqua	bird	√	
Coprosma robusta	bird	√	
Dacrydium cupressinum	bird		√
Dacrycarpus dacrydioides	bird	√	√
Hedycarya arboreus	bird	√	√
Macropiper excelsum	bird	√	√
Melicytus ramiflorus	bird	√	√
Myrsine australis	bird		√
Myrsine salicina	bird	√	√
Pittosporum colensoi	bird	√	
Pittosporum eugenioides	bird	√	
Prumnopitys ferruginea	bird		√
Pseudopanax arboreus	bird	√	
Pseudopanax crassifolius	bird	√	√
Rhopalostylis sapida	bird		√
Ripogonum scandens	bird		√
Schefflera digitata	bird		√
Hebe elliptica	wind	√	
Parsonsia heterophylla	wind	√	√
Weinmannia racemosa	wind	√	√

The presence of a woody species in the canopy was not found to be necessary in order for regenerating seedlings to appear within a study site (Table 3.8, Figure 3.6, Figure 3.7). Thirteen species were located in various planted restoration study sites as regenerating seedlings when the species was not present as part of the canopy. Two of these species were wind dispersed (*Parsonsia heterophylla* and *Weinmannia racemosa*).

Table 3.8: This table indicates study sites in which woody species were located, either as part of the canopy or as regenerating seedlings. C+ represents the presence of a species in the canopy, C- represents the absence of a plant in the canopy, and S+ and S- symbolise the respective presence or absence of a species as regenerating seedlings. For example, the fist column shows study sites in which a particular species was located in the canopy, but no evidence of its regeneration was detected.

Species	C+ S-	C- S+	C+ S+
Aristotelia serrata	P6, R3		P6
Carpodetus serratus	R2	P2, P4, P6	R1
Coprosma areolata	P4, R2	R3	
Coprosma grandifolia		P3, P4	P2, P5, P6, R1, R2, R3
Coprosma lucida	P5		
Coprosma propinqua	P1, P2, P4, P5, R1, R3	P3	
Coprosma propinqua/robusta	P1, P2, P4, P5		
Coprosma repens	P1, P2, P5		
Coprosma robusta	P1, P5, R2		P2, P3, P4
Cordyline australis	P1, P2, P4, P6		
Cortaderia richardii	P1, P2, P6		
Dacrydium cupressinum	R1		R2
Dacrycarpus dacrydioides		P2, P4, P6, R3	R1, R2
Dodonaea viscosa	P2		
Elaeocarpus dentatus	R2		
Freycinetia baueriana	R1, R2		
Hebe elliptica	P1, P2, P4, P5		P3
Hebe salicifolia	P1, P2, P5		
Hedycarya arborea		P2	R1, R2, R3
Leptospermum scoparium	P1, P2, P4, P5, P6		
Macropiper excelsum		P3	R3
Melicytus ramiflorus			P2, P3, P4, P6, R1, R2, R3
Metrosideros diffusa	R1, R2		
Metrosideros fulgens	R1, R2		
Metrosideros perforata	R1, R2		
Myoporum laetum	P1, P3		
Myrsine australis	R3	R1	
Myrsine salicifolia	P5	P6	R1, R2, R3
Olearia avicenniifolia	R2, R3		
Ozothamnus leptophyllus	P1		
Parsonsia heterophylla		P2, R1, R3	R2
Phormium cookianum	P2		
Phormium tenax	P1, P2, P3, P4, P5		
Pittosporum colensoi	P2, P4, P5		P3
Pittosporum crassifolium	P6		
Pittosporum eugenioides	P1, P2, P5	P3	P4
Pittosporum tenuifolium	P2, P4, P5, P6		
Prumnopitys ferruginea	R1		R2
Pseudopanax arboreus		P6	
Pseudopanax crassifolius		P2	R1, R2
Rhopalostylis sapida			R1, R3
Ripogonum scandens			R1, R2
Schefflera digitata	P3, P6, R2		R1
Weinmannia racemosa		P6	R1, R2

A one-way ANOVA run to investigate differences in the number of seedlings detected (regeneration density) in remnant and planted restoration study sites suggested a significant difference was present (F = 113.36, df = 1, P = < 0.01) According to the Tukey's test the mean density of seedlings found in remnant study sites was significantly greater than the density of seedlings present in planted restoration sites. A similar ANOVA run to detect statistically significant differences in regeneration density in planted restoration study sites indicated that a slight significant difference was present between the six study sites (F = 3.44, df = 5, P = < 0.01). The Tukey's test revealed that regenerating seedlings were found in significantly greater density in P6 than in P1 or P2.

Density of regeneration did not appear to be solely dependent on the dominance of the particular species within the canopy. Figure 3.6 illustrates a comparison of regeneration density with cover (using importance values) of species within planted restoration sites. Species located as seedlings with the greatest density in planted restoration sites, such as *Dacrycarpus dacrydioides* and *Coprosma grandifolia*, were not species present in the plantings at canopy level, indicating that sufficient fruiting species are planted within restoration study sites to attract bird species to stay for long enough periods for them to disperse fruit. Species that were planted, forming the shrub stratum in planted restoration sites, tended to be absent or poorly represented as regeneration. Similarly, within remnant sites, the importance of species identified as part of the canopy cover did not play an overriding role in influencing the abundance of seedlings on the forest floor. However, there was a tendency for species that were prevalent as part of the canopy layer to be located in greater densities as seedlings within remnant sites (Figure 3.7) than was the case for planted restoration study sites.

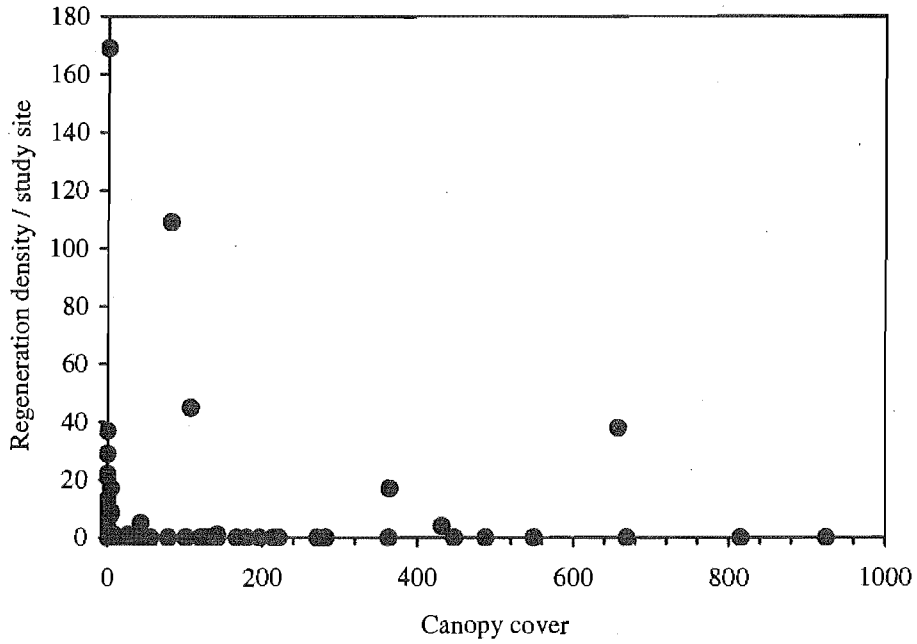


Figure 3.6: Comparison of regeneration density and cover of 'all woody species' within planted restoration study sites.

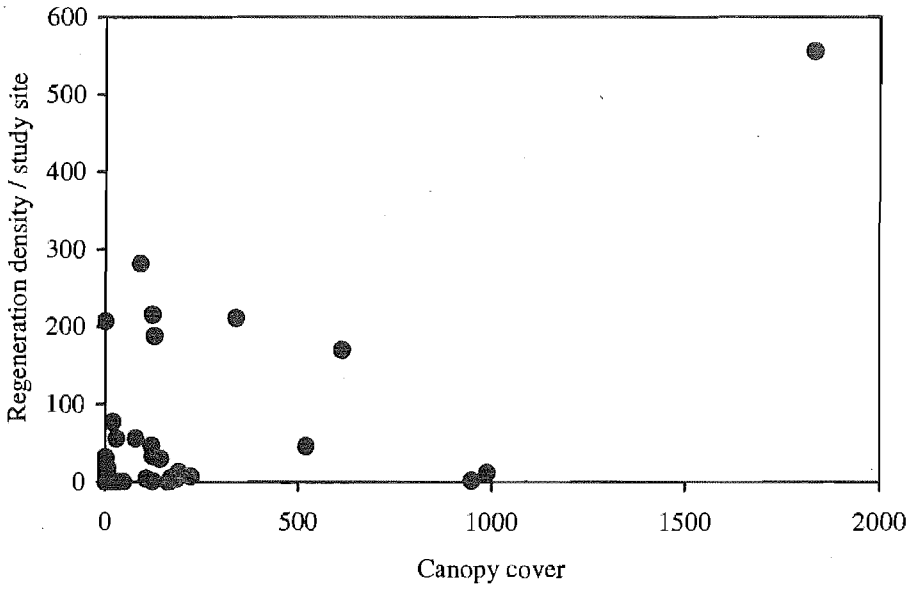


Figure 3.7: Comparison of regeneration density and cover of 'all woody species' within remnant study sites.

3.3.2. Ferns and Allied Plants

3.3.2.1. Diversity Assessments

Species Richness

A statistically significant difference was evident in the mean number of fern species present in remnant study sites compared with fern species in planted restoration sites (F = 536.90, df = 1, P = < 0.001). This significant difference detected by ANOVA was elaborated on by the Tukey's test, which showed that remnant sites had significantly more fern species present on average than planted restoration sites. Additionally, statistically significant differences were found in the mean number of fern species present in planted restoration sites (F = 19.81, df = 5, P = < 0.001). Tukey's 'comparison of means' test revealed that P6 had significantly more fern species present than all other planted restoration sites (Table 3.9).

Table 3.9: Descriptive statistics for species richness (S) of fern species in study sites. Part (a) investigates the difference between remnant and planted restoration sites, while part (b) compares species richness values between all six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	1.0	8.0	4.7^{A}	0.2
	P	120	0.0	4.0	0.4^{B}	0.1
(b)	P1	20	0.0	0.0	0.0^{B}	0.0
	P2	20	0.0	2.0	0.4^{B}	0.1
	P3	20	0.0	2.0	0.4^{B}	0.2
	P4	20	0.0	2.0	0.2^{B}	0.1
	P5	20	0.0	1.0	0.15^{B}	0.1
	P6	20	0.0	4.0	1.7^{A}	0.3

No fern species were located within study plots in P1. P3 contained the greatest total number of fern species, for planted restoration sites (S = 4). Remnant study sites were found to contain many more fern species than planted restoration sites (Figure 3.8a).

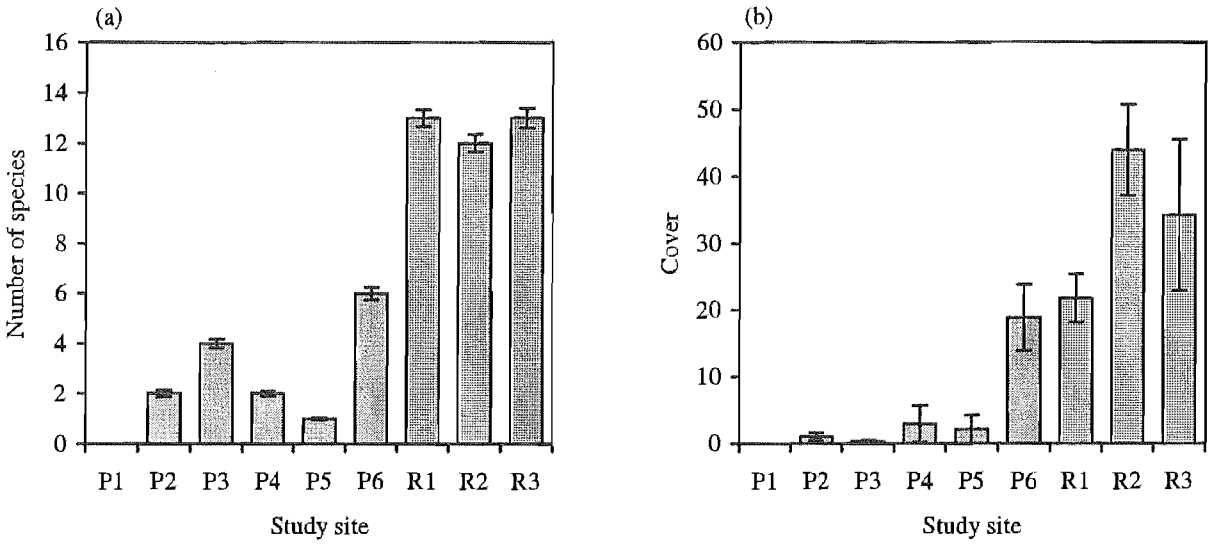


Figure 3.8: Graph (a) shows the number of different fern species in each study site; graph (b) illustrates the cover of ferns and allied plants in each study site, present over all vertical strata. The error bars depict the standard error around the mean for individual study sites.

Cover

Ferns and allied plants comprised a large portion of cover within remnant study sites (Figure 3.8b). Within planted restoration sites, ferns primarily occupied space <0.5 m in height, whereas in the three remnant sites, the vast majority of fern species were found to be present in vertical strata above 0.5 m in height (i.e. as tree ferns). 91.1%, 89.1%, and 95.8% of fern species occurred in canopy strata in R1, R2 and R3, respectively. P6 had the greatest amount of cover represented by fern species out of the six planted restoration sites (18.8), while fern species were not located within the P1 study site. R2 had the greatest mean value for cover by fern species overall (43.9).

The results of a one-way ANOVA for cover by type of study site (remnants or planted restoration sites) showed that a significant difference existed ($F = 75.00$, df = 1, $P = < 0.001$). According to the Tukey's test, remnant sites had significantly more fern cover than did planted restoration sites. Cover represented by fern species was found to differ significantly between planted restoration sites ($F = 7.16$, df = 5, $P = < 0.001$) (Table 3.10). The Tukey's test revealed that the P6 study site had a larger mean area

of cover represented by ferns and allied plants than all other planted restoration study sites.

Table 3.10: Descriptive statistics for cover of fern species (using importance values) in study sites. Part (a) investigates the difference in cover represented by fern species in remnants and planted restoration sites, while part (b) compares cover values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.0	156.5	33.3^{A}	4.6
	P	120	0.0	50.6	3.1^{B}	0.9
(b)	P1	20	0.0	0.0	0.0^{B}	0.0
	P2	20	0.2	3.8	1.2^{B}	0.6
	P3	20	0.0	2.9	0.3^{B}	0.2
	P4	20	0.0	55.0	2.9^{B}	2.8
	P5	20	0.0	42.0	2.1^{B}	2.1
	P6	20	0.0	93.5	18.8^{A}	4.9

Heterogeneity

An ANOVA revealed mean values of *H'* were significantly different in remnant sites than in planted restoration sites (F = 206.70, df = 1, P = < 0.001). According to the Tukey's test, remnant sites were significantly more heterogeneous than planted restoration sites, in terms of fern species. Significant differences for *H'* were also found between planted restoration sites (F = 6.59, df = 5, P = < 0.001). Tukey's 'comparison of means' test was run to determine the nature of these differences. This test revealed that P6 was significantly more heterogeneous than P2, P4, P1 and P5. The P3 study site did not differ significantly from P6 in terms of *H'* values (Table 3.11).

All mean values for the Shannon index for fern species lay below the lower boundary of the range of values stated by Magurran (1988), as that within which the majority of data sets lie. P6 had the highest value over all study sites (*H'* = 0.26). P1 and P5 had zero values for this heterogeneity index. R3 was the most heterogeneous study site (*H'* = 0.83) (Figure 3.9a).

Table 3.11: Shannon's diversity index (H') for fern species. Part (a) compares H' values in remnant and planted restoration sites, while part (b) shows H' values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.00	1.73	0.77^{A}	0.06
	P	120	0.00	1.10	0.07^{B}	0.02
(b)	P1	20	0.00	0.00	0.00^{B}	0.00
	P2	20	0.00	0.69	0.03^{B}	0.03
	P3	20	0.00	0.69	0.10^{AB}	0.06
	P4	20	0.00	0.39	0.02^{B}	0.02
	P5	20	0.00	0.00	0.00^{B}	0.00
	P6	20	0.00	1.10	0.26^{A}	0.07

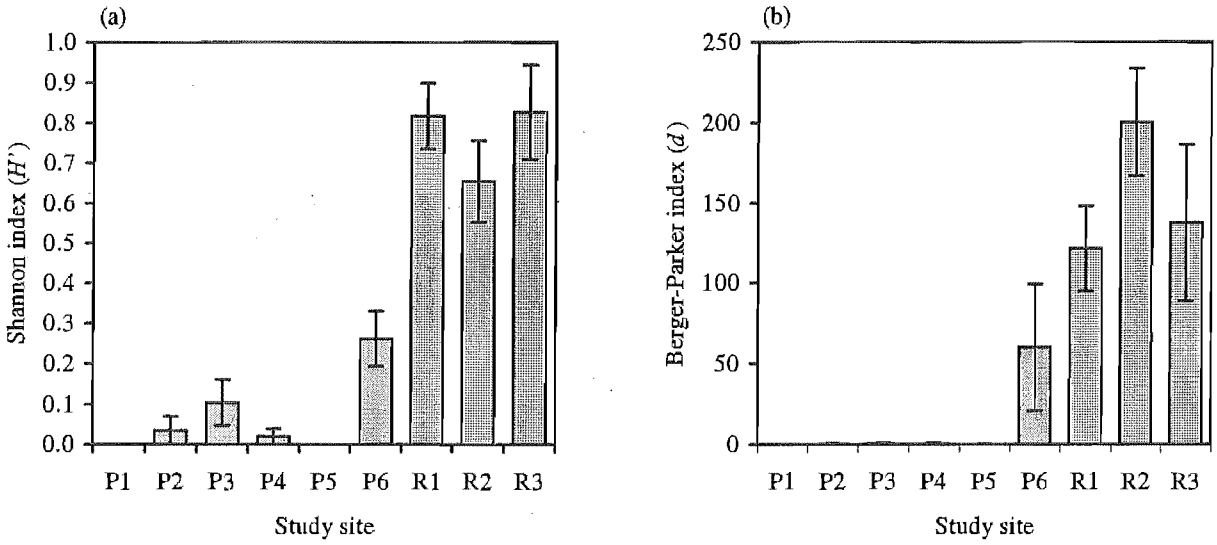


Figure 3.9: Graphs (a) and (b) illustrate the relationship of fern and allied plants between study sites using two heterogeneity indices: Shannon's index (H') and the Berger-Parker index (d), respectively. Error bars are the standard error around the mean.

Dominance

R2 had the greatest value for the reciprocal of the Berger-Parker index (d = 200.25), indicative of the lack of dominance of one fern species (Figure 3.9b). P1 had a zero value for this dominance index, as no fern species were located. P6 had the greatest value (d = 59.90) for planted restoration sites, indicating the high diversity and lack of dominance of one particular fern species within this study site. However, values for the Berger-Parker index varied greatly in this study site (Table 3.12).

The results of an ANOVA revealed that statistically significant differences existed between planted restoration and remnant sites with regards to dominance of fern species (F = 62.83, df = 1, P = < 0.001). According to the Tukey's test the mean d value for remnant study sites was significantly greater than the mean d value obtained for planted restoration sites. This smaller d value for planted restoration sites results form the smaller number of fern species present, hence the dominance of one or a few fern species. It was suggested by the results of a one-way ANOVA for dominance values by individual restoration sites that a slight statistically significant difference was present between the six study sites (F = 2.32, df = 5, P = 0.048). However, when multiple comparison tests were run using the Tukey's test, the nature of these differences was not exposed, possibly due to the high variance associated with the Berger-Parker index in P6. An alternative multiple comparison test was used to investigate the differences. The LSD test showed that P6 had significantly greater dominance value than all other planted restoration sites, representing the lack of dominance of one particular fern species within this study site.

Table 3.12: Descriptive statistics of the Berger-Parker index (d) for fern species. Part (a) of the table looks at descriptive statistics for remnants and planted restoration sites, while (b) shows values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	1.00	669.56	152.99^{A}	21.70
	P	120	0.00	785.17	10.25^{B}	6.70
(b)	P1	20	0.00	0.00	0.00^{A}	0.00
	P2	20	0.00	4.00	0.45^{A}	0.21
	P3	20	0.00	4.00	0.65^{A}	0.33
	P4	20	0.00	0.86	0.48^{A}	0.43
	P5	20	0.00	1.00	0.05^{A}	0.05
	P6	20	0.00	785.17	59.90^{A}	39.12

Evenness

P6 had the greatest evenness of fern species within planted restoration sites (E = 0.37), while R3 was the most even of all study sites (E = 0.51). P1 and P5 had zero values for this diversity index (Figure 3.10). A statistically significant difference was present between planted restoration and remnant sites in terms of the modified Hill's ratio for evenness, as revealed by a one-way ANOVA (F = 123.63, df = 1, P = < 0.001). Remnant sites had significantly more even distribution of fern species than did

planted restoration sites. Similarly, evenness values were found to be significantly different between individual planted restoration sites ($F = 7.47$, df = 5, $P = < 0.001$). The Tukey's test elaborated on this difference, showing that P6 had significantly higher average evenness values than all other planted restoration study sites (Table 3.13).

Table 3.13: The modified Hill's ratio for evenness (E) of fern species. Part (a) of the table investigates differences in E for remnants and planted restoration sites, while (b) compares E values for all six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.00	1.00	0.54^{A}	0.03
	P	120	0.00	1.00	0.10^{B}	0.02
(b)	P1	20	0.00	0.00	0.00^{B}	0.00
	P2	20	0.00	1.00	0.05^{B}	0.05
	P3	20	0.00	1.00	0.15^{B}	0.08
	P4	20	0.00	0.63	0.03^{B}	0.03
	P5	20	0.00	0.00	0.00^{B}	0.00
	P6	20	0.00	1.00	0.37^{A}	0.08

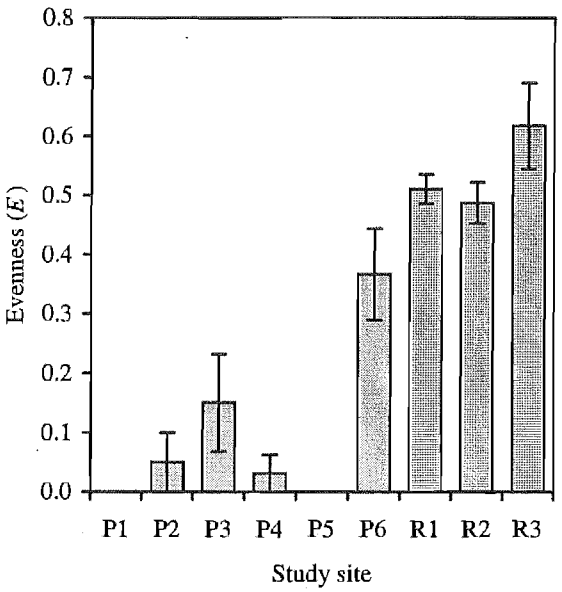


Figure 3.10: Comparison of mean values for Hill's modified ratio for evenness (E) for ferns and allied plants between the nine study sites. Error bars are the standard error around the mean.

Floristic Similarity

Comparison of fern species present in study sites resulted in numerous zero values for Jaccard's coefficient (Table 3.14). R2 and R3 were the study sites that shared the most fern species in common (C_J = 0.79). Fern species present in all remnant sites and those fern species present in planted restoration sites had a similarity coefficient of C_J = 0.375.

Table 3.14: Jaccard's coefficients for floristic similarity of fern species between study sites.

	P1	P2	P3	P4	P5	P6	R1	R2	R3
P1	0.00								
P2	0.00	1.00							
P3	0.00	0.20	1.00						
P4	0.00	0.00	0.50	1.00					
P5	0.00	0.00	0.25	0.50	1.00				
P6	0.00	0.33	0.67	0.33	0.17	1.00			
R1	0.00	0.07	0.31	0.15	0.08	0.19	1.00		
R2	0.00	0.17	0.23	0.08	0.00	0.38	0.67	1.00	
R3	0.00	0.15	0.31	0.15	0.08	0.46	0.63	0.79	1.0

3.3.3. All Woody Species

3.3.3.1. Diversity Assessments

Species Richness

The results of the one-way ANOVA for species richness by type of site (planted restoration or remnant sites) found that there was a significant difference between the two types of study sites (F = 92.64, df = 1, P = < 0.001). The Tukey's test revealed that remnant sites had significantly more woody species present, on average, than did planted restoration sites. Additionally, the ANOVA suggested significant differences existed between mean species richness for different aged planted restoration sites (F = 6.97, df = 5, P = < 0.001). P1 had the greatest mean number of species present and this value, according to the Tukey's test, was significantly greater than the mean number of species present in P4, P6 and P5. P2 had a significantly higher mean number of species than P5 and P6, as did P3 (Table 3.15).

Table 3.15: Species richness (*S*) for all woody species. Part (a) investigates the difference between remnant sites and planted restoration sites, while (b) compares species richness values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	1.0	10.0	5.8^{A}	0.3
	P	120	0.0	7.0	3.0^{B}	0.1
(b)	P1	20	1.0	7.0	4.1^{A}	0.4
	P2	20	2.0	6.0	3.5^{AB}	0.2
	P3	20	1.0	6.0	3.4^{AB}	0.4
	P4	20	0.0	7.0	2.7^{BC}	0.4
	P5	20	0.0	4.0	2.0^{C}	0.2
	P6	20	1.0	4.0	2.3^{C}	0.2

The variety of woody species present in planted restoration sites was primarily dictated by the number of different species planted. P3 had the greatest total species richness for restoration sites, with eighteen different tree or shrub species present over the whole study site. P5 had the lowest species richness, with just seven species located in the shrub layer of the study site. The three remnant sites varied in the total number of different woody species present, from twelve species in R3 to twenty-one present in R2 (Figure 3.11a).

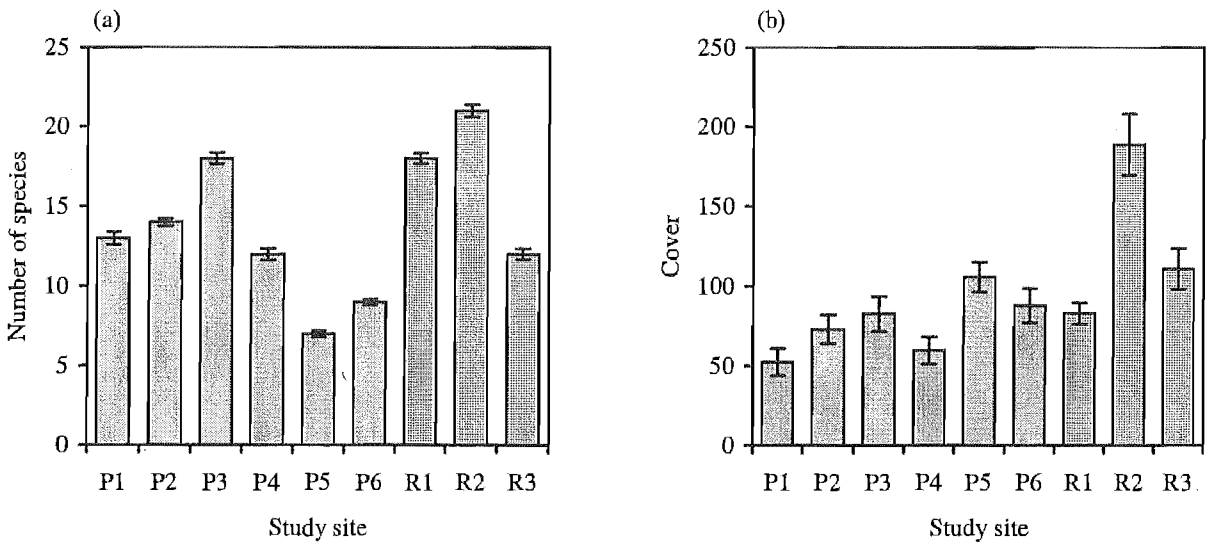


Figure 3.11: Graph (a) shows the species richness (*S*) of each site for all woody species, while graph (b) illustrates total cover represented by woody species, within combined canopy levels, for all nine study sites. Error bars are the standard error around the mean number of species and cover for each study site.

Cover

The amount of vegetation cover provided by woody species was found to be quite varied between study sites (Figure 3.11b). In planted restoration sites, P5 had the greatest amount of cover (105.6). P1 had the lowest cover value for woody species (52.3). R2 had the greatest amount of cover by woody species for all study sites (188.7). Cover of woody species within planted restoration sites was present in a shrub stratum only. In remnant sites however, woody vegetation was found over three vertical strata: shrub, sub-canopy, and canopy. R3 often lacked a distinct sub-canopy layer. All three remnant study sites had the greatest amount of cover provided by tree and shrub species in the actual canopy strata (53.2%, 65.5%, and 86.6% for R1, R2 and R3, respectively). R1 and R2 had a sub-canopy layer of woody vegetation, comprising 24.0% and 23.9% of total cover, respectively. Shrub strata contained 22.8% of total cover in R1, 10.6% in R2, and 13.4% in R3.

An ANOVA indicated that a significant difference was present between cover represented by woody species in planted restoration sites and cover of woody species in remnant sites (F = 31.07, df = 1, P = < 0.001). The cover represented by woody species was shown to be significantly greater in remnant sites than cover of woody species in planted restoration sites by the Tukey's test. The results of an ANOVA for cover by planted restoration sites indicated that significant differences were present between the amount of cover represented by woody species and different planted restoration study sites (F = 4.12, df = 5, P = 0.002). According to the Tukey's test P5 had significantly greater cover represented by woody species than did P4 or P1 (Table 3.16).

Table 3.16: Descriptive statistics of values for total cover of all woody species in study sites. Part (a) looks at descriptive statistics for remnants and planted restoration sites, while (b) details cover values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	10.4	368.4	127.4^{A}	9.8
	P	120	0.0	163.2	76.8^{B}	4.2
(b)	P1	20	5.2	124.9	52.3^{B}	8.5
	P2	20	16.3	161.0	72.9^{AB}	9.0
	P3	20	11.8	163.2	82.6^{AB}	11.0
	P4	20	0.0	127.0	59.8^{B}	8.5
	P5	20	0.0	149.2	105.6^{A}	9.4
	P6	20	4.4	146.2	87.7^{AB}	10.8

Heterogeneity

A significant difference was detected by ANOVA for mean values of Shannon's diversity index (*H'*) between remnant and planted restoration sites ($F = 10.80$, df = 1, $P = 0.001$). Tukey's test was used to elaborate on this difference, revealing that planted restoration sites had on average significantly smaller *H'* values, although P1 had the highest overall value. The results from ANOVA for testing *H'* by planted restoration study sites indicated that differences in mean values existed between the six study sites ($F = 5.46$, df = 5, $P = < 0.001$). In order to determine the nature of these differences Tukey's test was run. P1 was found to have significantly higher mean *H'* values than P5 and P6. P2 was also significantly more heterogeneous than P5 and P6 (Table 3.17).

P1 was found to contain the most diverse array of woody species present ($H' = 1.00$) in planted restoration sites, while P6 comprised the least heterogeneous site for all study sites ($H' = 0.45$) (Figure 3.12a). R1 was the most diverse study site overall, with an *H'* value of 1.45.

Table 3.17: Descriptive statistics of values for the Shannon index (*H'*) for 'all woody species'. Part (a) looks at descriptive statistics for remnants and planted restoration sites, while (b) denotes *H'* values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.00	1.89	0.98^{A}	0.08
	P	120	0.00	1.95	0.72^{B}	0.04
(b)	P1	20	0.00	1.95	1.00^{A}	0.13
	P2	20	0.28	1.49	0.93^{A}	0.08
	P3	20	0.00	1.58	0.76^{AB}	0.11
	P4	20	0.00	1.59	0.69^{AB}	0.10
	P5	20	0.00	1.10	0.47^{B}	0.08
	P6	20	0.00	1.04	0.45^{B}	0.06

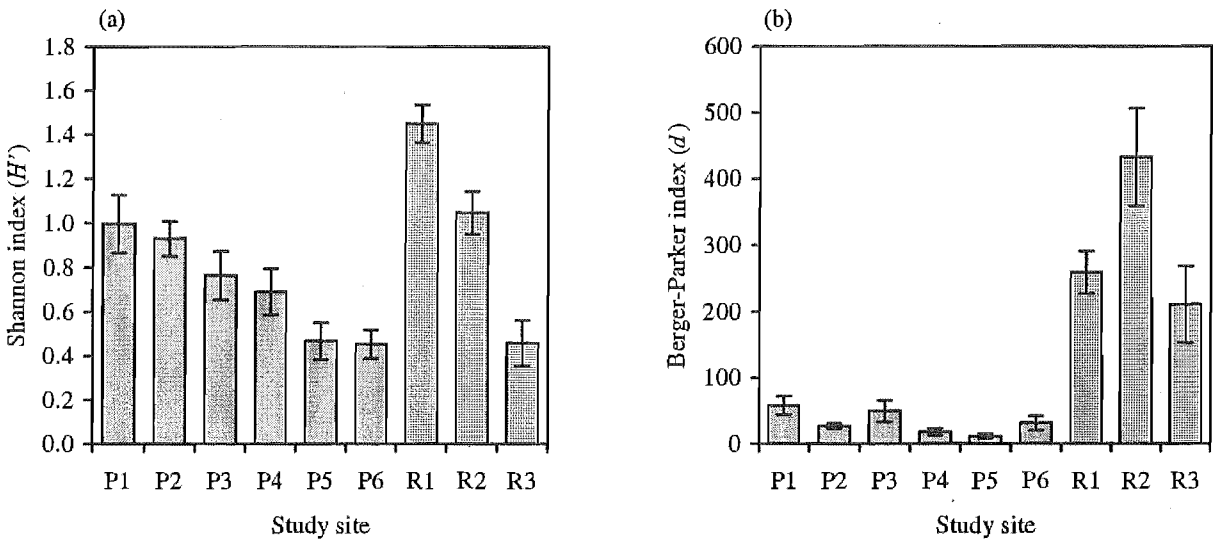


Figure 3.12: Graph (a) shows the relationship between study sites and the Shannon index (*H'*), while graph (b) illustrates the relationship between study sites using the Berger-Parker index (*d*) for 'all woody species'. Error bars are the standard error around the mean for both graphs.

Dominance

Mean values obtained for the Berger-Parker index (*d*) differed significantly between planted restoration and remnant sites (F = 112.49, df = 1, P = < 0.001). The Tukey's test showed that remnant sites had significantly higher mean values for *d*, indicative of the lack of dominance of a few woody species in such sites. ANOVA results for the dominance index by planted restoration study site suggested that there were significant differences between planted restoration study sites (F = 3.11, df = 5, P = 0.011). According to the Tukey's test, mean values of *d* were significantly greater in P1 than P5 (Table 3.18). R2 obtained the highest value for the Berger-Parker index (*d* = 432.32), suggestive of the high diversity of woody species present in the site. P5 had the lowest overall value for this dominance index (*d* = 11.15), indicating that woody species present in this site were dominated by a few particular species (Figure 3.12b). P1 was the most heterogeneous of the planted restoration sites with a dominance value of 58.17.

Table 3.18: Descriptive statistics of values for the Berger-Parker index (*d*) for 'all woody species'. Part (a) looks at descriptive statistics for remnants and planted restoration sites, while (b) shows values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	1.00	1241.19	300.61^{A}	34.75
	P	120	0.00	336.16	32.44^{B}	4.40
(b)	P1	20	1.00	225.73	58.17^{A}	14.60
	P2	20	5.04	77.89	26.68^{AB}	4.67
	P3	20	1.00	336.16	49.59^{AB}	16.18
	P4	20	0.00	96.62	17.80^{AB}	5.34
	P5	20	0.00	63.73	11.15^{B}	3.41
	P6	20	1.00	203.20	31.26^{AB}	10.89

Evenness

An ANOVA run to determine the difference for mean evenness values by type of study site (planted restoration or remnant sites) demonstrated that no significant difference was present between the two types of study site (F = 1.22, df = 1, P = 0.271). Similarly, mean values for the modified Hill's ratio for evenness between the six planted restoration sites were not found to differ significantly (F = 1.73, df = 5, P = 0.134) (Table 3.19). P2 had the highest value for the modified Hill's ratio for evenness (E = 0.77). P6 had the lowest evenness value for the restoration plantings (E = 0.58), while R3 was found to have the lowest evenness value overall (E = 0.47) (Figure 3.13).

Table 3.19: The modified Hill's ratio for evenness (*E*) for 'all woody species'. Part (a) looks at descriptive statistics for remnants and planted restoration sites, while part (b) compares evenness values for all six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	60	0.00	0.94	0.59^{A}	0.03
	P	120	0.00	1.00	0.64^{A}	0.03
(b)	P1	20	0.00	1.00	0.72^{A}	0.06
	P2	20	0.43	0.95	0.77^{A}	0.03
	P3	20	0.00	0.95	0.62^{A}	0.07
	P4	20	0.00	1.00	0.66^{A}	0.08
	P5	20	0.00	1.00	0.59^{A}	0.09
	P6	20	0.00	0.94	0.58^{A}	0.07

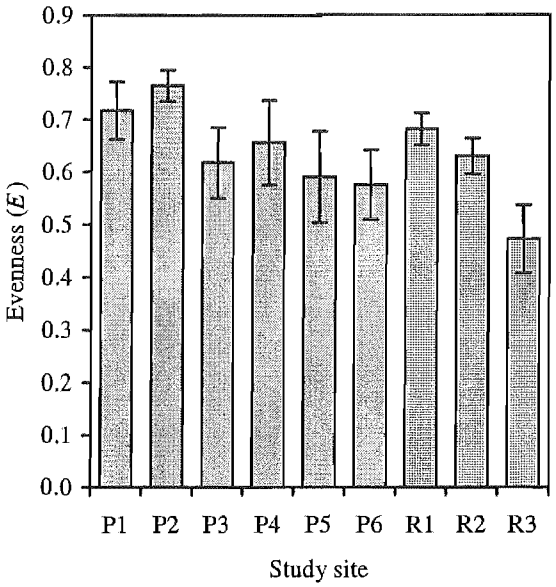


Figure 3.13: Results of the modified Hill's ratio for evenness (E) of 'all woody species' for each study site. Error bars are the standard error around the mean.

Floristic Similarity

Remnant and planted restoration study sites shared very few species in common (C_J = 0.19). The most similar study sites in terms of species present were P2 and P3 (C_J = 0.68). P1 was the most disparate from remnant sites R1 and R2 as based on species presence (C_J = 0.03), and R3 (C_J = 0.04) (Table 3.20).

Table 3.20: Jaccard's coefficients of floristic similarity for 'all woody species' between study sites.

	P1	P2	P3	P4	P5	P6	R1	R2	R3
P1	1.00								
P2	0.50	1.00							
P3	0.55	0.68	1.00						
P4	0.47	0.53	0.58	1.00					
P5	0.25	0.24	0.25	0.36	1.00				
P6	0.16	0.15	0.29	0.27	0.14	1.00			
R1	0.03	0.07	0.09	0.07	0.09	0.13	1.00		
R2	0.03	0.09	0.11	0.10	0.12	0.11	0.70	1.00	
R3	0.04	0.13	0.15	0.09	0.12	0.17	0.36	0.27	1.00

3.3.3.2. Growth in Planted Restoration Study Sites

For all species combined, average height increases occurred in a manner consistent with increasing time since planting (Table 3.21) up until the P5 study site. P6 had less average percentage height increase over all species than did P5. P6 may have had lesser height increases due to possible lack of post-planting maintenance, resulting in plantings having more difficulty establishing initially, thereby slowing growth overall. An estimation of the percentage height increase for species within planted restoration sites is included in Table 3.22 for selected species present in at least four different study sites.

Table 3.21: Estimated percentage increase in height for all planted species combined, in planted restoration study sites.

Study site	Average Growth (%) ± SE
P1	260.0 ± 18.6
P2	593.6 ± 63.0
P3	825.4 ± 80.5
P4	871.5 ± 115.5
P5	1249.1 ± 101.8
P6	910.4 ± 164.2

Table 3.22: Estimated percentage height increase of woody species within planted restoration study sites. Species listed were ones that were located within at least four different planted restoration study sites. The symbol – indicates the absence of the particular species from a study site.

	Planted Restoration Study Site					
Species	**P1**	**P2**	**P3**	**P4**	**P5**	**P6**
Coprosma propinqua	278.0	995.9	1380.3	1610.0	–	–
Coprosma robusta	263.3	617.3	906.7	1053.9	978.9	–
Cordyline australis	174.8	–	768.6	763.3	–	1166.7
Hebe elliptica	304.4	620.3	980.7	672.2	1207.2	–
Leptospermum scoparium	207.7	786.8	658.8	766.7	–	1618.4
Phormium tenax	276.8	389.0	515.1	680.0	1363.6	–
Pittosporum colensoi	–	645.0	1187.5	1386.2	1250.0	–
Pittosporum eugenioides	231.5	544.0	795.0	1166.7	–	–
Pittosporum tenuifolium	–	230.0	185.0	560.0	–	1704.5

3.4. Preliminary Discussion of Results

This preliminary discussion of results is limited in scope to consideration of the results contained within this chapter only. The section has been divided into five subheadings that correspond to the objectives outlined at the beginning of this chapter (Section 3.1.). Discussion of these results with regard to how they sit with other aspects of this study will be undertaken in Chapter 6, in addition to their impact on initial restoration success.

Numerous study plots were established within each study site in order to compare vegetation patterns between sites. It is acknowledged that small sample sizes tend to underestimate differences between study sites. However the number of study plots used, twenty 4 × 4 m plots per study site, is thought to fairly reflect both the composition and abundance of the sites from which samples were obtained (Cao et al, 2002). Diversity assessments and statistical analyses were used to illustrate floristic differences between vegetation composition between the six planted restoration sites, and particularly between planted restoration and remnant study sites.

3.4.1. Did Species Composition Vary Between Planted Restoration and Remnant Study Sites?

3.4.1.1. Regeneration

Diversity assessments highlighted differences between regeneration patterns in remnant and planted restoration study sites. There was a tendency for planted restoration study sites to have less regeneration than remnants, resulting in less cover represented by seedlings. Additionally, regenerating seedlings within planted restoration study sites tended to be less heterogeneous, and subsequently were dominated by fewer species as well as having a significantly less even spread of regenerating seedlings than remnant study sites.

Reay & Norton (1999a) found, in their study of restoration success at Kennedy's Bush, Canterbury, regenerating vegetation within three restoration study plots "surprisingly similar to that of the natural regeneration and mature forest sites" (p.304). They linked the vegetative similarity of regenerating seedlings in their study

to the ability of planted species to facilitate the recolonisation and establishment of similar regenerating species that would be expected to occur during natural succession (Hobbs & Norton, 1999a). The disparity of number of spread of regenerating species found in planted restoration and remnant study sites in this study is attributed to the young age, particularly in comparison to Hobbs & Norton (1999a), of the restoration plantings and the subsequent paucity of suitable perch sites for birds.

A variety of species of regenerating woody species were detected within remnant study sites. Significantly more species were present in this type of study site compared with the number of species of regenerating seedlings located in planted restoration sites. Conditions suitable to facilitate regeneration, such as appropriate light conditions, may have been more prevalent within remnant study sites (e.g. because of greater horizontal and vertical heterogeneity). Where regeneration was observed within planted restoration sites it was almost exclusively beneath plantings. As a consequence, the establishment of adequate canopy cover was assumed to be a key driver enabling the establishment of seedlings. It can further be inferred from this observation that exotic grass growth within planted restoration study sites was stifling the ability of seedlings to germinate and establish at these locations. Smale et al (2001) recognised mean aspect and slope as factors that may affect soil moisture regimes and consequently seedling establishment. Ground litter was found to be a key influence on the presence of regenerating seedlings (Chapter 5).

An alternative explanation for the paucity of regenerating seedlings witnessed away from the shelter of a planted species is that the grass growth made it difficult to encounter examples of regeneration, although this would seem a less plausible explanation. Appearance of regeneration does not indicate that seedlings encountered will necessarily survive to form part of the canopy. However, the inference is that if environmental conditions are suitable to allow woody species to germinate then the possibility of species establishment and continued growth is promising.

The density of regeneration was found to be significantly greater in remnant study sites than planted restoration study sites. This difference in density may be the result of the occurrence of regenerating seedlings from canopy species within remnant sites,

whereas many regenerating species require dispersal into planted restoration sites. Alternatively, seeds may be dispersed into planted restoration sites, but fail to germinate due to environmental factors such as competition from a dense grass sward, or a lack of suitable light or microclimatic conditions.

3.4.1.2. Ferns and Allied Plants

Significantly more fern species were identified within study plots in remnant sites than in planted restoration study sites. This pattern occurred when investigating the cover represented by fern species also. The ability of fern species to flourish within remnant study sites may be the result of more favourable moisture and light conditions produced due to the closed canopy. Within remnant study sites, fern species were present at ground level (< 0.5 m) and in higher vertical strata (> 0.5 m above ground level). Not surprisingly, comparison of the floristic similarity revealed that study sites shared few fern species in common, excepting remnant study sites.

3.4.1.3. All Woody Species

Significantly more woody species (excluding regenerating seedlings <0.3 m in height) were found as part of the canopy stratum in remnants than in planted restoration study sites. Study sites with greater species richness may indicate more competitive environments, which in turn restrict the ability for potentially invasive species to inhabit, providing exogenous factors do not vary between the sites (Naeem et al, 2000). The number of woody species present within planted restoration study sites was a direct result of the variety of species planted.

Mean cover of woody species was also found to be significantly greater within remnant sites than in planted restoration study sites. This greater cover of species in remnant sites may be the result of small disturbances, such as windfall, providing gaps into which various woody species were able to establish, thereby increasing the amount of cover of woody species (Wells et al, 1998). Mean cover of woody species was greater in remnant study sites due to the greater vertical complexity reflecting increasing stand age. The oldest planted restoration study site used (P6) was planted in 1980 whereas, although not undisturbed, remnant study sites have been extant for considerably more time. Species of climbers were included within the 'all woody species' category. Climbers were only located (with the exception of regenerating

seedlings) within remnant study sites, an additional source of the comparatively high cover values.

The vegetation of remnant study sites was found to be significantly more heterogeneous than in planted restoration study sites. This result was not unexpected, following the similar pattern detected in the mean number of different species present in each type of study site. Reay & Norton (1999a) found, when assessing restoration success of plantings in the Port Hills, Canterbury, New Zealand, that levels of compositional heterogeneity were similar between the two oldest planted study sites and naturally regenerating and mature forest sites. The restoration sites used were 30 and 35 years old, providing a possible indication of the time span necessary to achieve planted restoration study sites with similar vegetation composition to the remnant study sites in this study. Compositional heterogeneity however, was the only structural aspect for which Reay & Norton (1999a) found restoration sites to be similar to the naturally regenerating and mature forest sites.

3.4.1.4. Summary

Species composition of regenerating seedlings was found to vary significantly between planted restoration and remnant study sites. Planted restoration sites tended to have less regeneration, be less heterogeneous, and were dominated by fewer species than remnant study sites. The development of suitable conditions such as light, moisture and ground litter (as discussed in Chapter 5), particularly through the development of a continuous canopy cover, were seen to be important factors in the colonisation and establishment of regenerating seedlings.

3.4.2. Did Species Composition Vary Between Planted Restoration Study Sites?

3.4.2.1. Regeneration

The number of species of regenerating seedlings was found to vary significantly between planted restoration study sites. Minimal evidence of regeneration in the youngest two planted restoration sites was found, indicating that environmental conditions were taking some time to become suitable to facilitate regeneration. No regenerating seedlings were witnessed in the P1 study plot, which had only been

planted three years prior to this study. The lack of suitable perch sites within these two study sites may be influential in determining the presence of regenerating seedlings. Additionally, the two youngest sites have been established on overburden substrate, compared with pasture for the majority of the planted restoration sites.

An exception to the depauperate evidence of regenerating seedlings within planted restoration study sites was within P6, in which regeneration was found to be significantly more prevalent than in other planted restoration sites. A similar explanation to that given regarding the presence of fern species is appropriate for this site. Due to the markedly different vegetation structure in the 'oldest' planted restoration study site (P6), microclimatic conditions appropriate for seedling establishment were present allowing for the establishment and growth of seedlings for a considerably longer period of time than those available in other planted restoration sites.

Seedlings found in P5 were from species present in the canopy of this site indicating that pollination was occurring. Study sites P2, P3, P4, and P6 contained evidence of regenerating seedlings of species present in the shrub layer of these study plots in addition to species not present as part of the canopy (i.e. dispersal of novel species into these four study sites had occurred). The presence of species *Dacrycarpus dacrydioides* and *Coprosma grandifolia* also shows that the planted restoration sites are facilitating dispersal, as this species requires bird dispersal to reach these sites. The capacity for planted restoration study sites to facilitate colonisation and establishment of regenerating species is discussed in more detail in Section 3.4.3 and Section 3.4.4.

3.4.2.2. Ferns and Allied Plants

Where the sparse examples of ferns were found within planted restoration study sites, they were located < 0.5 m above the ground. Significantly more ferns occurred within P6 than other planted restoration study sites. Time lapsed since planting of the P6 study site and the resultant difference in vegetation structure may have been the cause of the greater variety of fern species detected here. This pattern continued for the three

β-diversity indices (the Shannon index, Berger-Parker index, and the modified Hill's ratio for evenness).

The presence of a fern understory has been found to influence the microenvironment of the forest floor, including light levels and litter layer, thereby influencing seedling emergence and establishment (George & Bazzaz, 1999a). George & Bazzaz (1999a) suggest that a fern understory may act as a selective filter influencing forest composition by reducing density, altering species composition, and determining the spatial composition of the seedling bank. The fern understory can also influence the growth and survival of tree seedlings by influencing the activity of invertebrate herbivores and predators (George & Bazzaz, 1999b).

3.4.2.3. All Woody Species

Statistically significant differences in the mean species richness of planted restoration sites were detected. No clear trend for the number of species to vary with increasing time since planting was evident. As stated previously, the number of woody species present within planted restoration sites was a direct subset of the variety of species planted. The P3 study site had the greatest total number of woody species (S = 18), while a mere five woody species were detected within the canopy strata in P5. This broad difference in species richness reflects the variation in the number of woody species propagated and grown to specification during different planting periods. Despite benefits of increased diversity, functional properties of species used in planted restoration sites more than species richness per se, affect ecosystem properties such as nutrient uptake (Hooper & Vitousek, 1997).

Total cover of woody species tended to increase with time since planting for the first three planted restoration study sites (P1, P2, and P3). Standard errors often overlapped for cover values (see Figure 3.2b); inferring interpretation of results must be undertaken with caution. Cover of woody species within the P4 study site was less than all other planted restoration sites, with the exception of P1, the most recently planted study site. The low cover value within P4 may be explained by the limited post-planting maintenance within the study site, which allowed the dense grass sward to overcome recent plantings, stifling their ability to grow or survive. Woody

vegetation within the P4 study site was arranged in a clumped fashion, suggestive of possible die-off of initial plantings. Average cover of woody species was found to be significantly greater in P5 than in P4 or P1. This difference can be explained by the difference in time lapsed since planting for P5 and P1.

Disparity between values of the Shannon index within planted restoration study sites was evident. A tendency for heterogeneity to progressively decrease with increasing time since planting was found. A few woody species dominated all six planted restoration study sites, as reflected in the low values recorded for the reciprocal of the Berger-Parker index. Similar to species richness values however, heterogeneity and species dominance within these six study sites was a direct result of planting pattern. Evenness did not vary significantly between the six planted restoration sites. This suggests that similar approaches to planting pattern were used within all study sites. Low floristic similarity of 'all woody species' was found for comparisons of most study sites. Only five comparisons of study sites were found to share more than half their woody species in common. An additional factor to consider with this lack of similarity between study sites is that Jaccard's coefficients are obtained by comparing presence-absence data of species, not taking species abundance into account.

Study of vegetation dynamics is common in restoration programmes (Parrotta & Knowles, 1999). Revegetation dynamics of cliff faces in abandoned limestone quarries in southern Ontario, Canada were studied by Ursic et al (1997). In this study, site age was found to be an important factor, in addition to the development of a dense tree canopy on the quarry floor, in influencing species assemblages on quarry walls.

3.4.2.4. Summary

Vegetation composition did vary between the six planted restoration study sites. Such variation at the shrub stratum is somewhat obvious to emphasise however, as compositional differences were primarily due to the number and type of species planted during the establishment of the different study sites. Variation in the composition of regeneration throughout the planted restoration sites was apparent, reflecting the development of suitable microhabitat conditions to facilitate dispersal and germination of seedlings.

3.4.3. Was Regeneration Dependent on the Dominance of Canopy Species?

Neither the presence of regenerating seedlings, nor their density appeared to be wholly dependent on the dominance of particular species within the canopy. The presence of regenerating seedlings was not found to be dependent on the dominance of canopy species. Twenty four different woody species were found as regenerating seedlings (as detailed in Table 3.20), fourteen of which were detected in study sites when the species did not form part of the canopy (in six of the nine study sites). Seedlings detected within planted restoration study sites with the greatest densities, such as *Dacrycarpus dacrydioides*, *Carpodetus serratus*, and *Coprosma grandifolia*, were not planted, indicating that dispersal of novel species into planted restoration study sites occurred. In terms of density, planted woody species were poorly represented as seedlings. This trend was also apparent in remnant study sites, for which dominance in the canopy cover did not play an overriding role in influencing the abundance of seedlings. A tendency for regeneration density to decrease with increasing canopy cover was apparent in both planted restoration and remnant study sites. The apparent outlier in Figure 3.13 was a legitimate result of the overwhelming dominance of *Coprosma grandifolia* as both a seedling and as part of the canopy in the R3 study site.

The results of a study by Parrotta (1995) suggested that canopy species exerted a significant influence on colonisation patterns of secondary forest species. Therefore, species planted in the restoration sites are an important consideration in the facilitation of regeneration of appropriate woody species. However, it is apparent from this study that the formation of a canopy cover, providing perch sites for birds, rather than the specific identity of species comprising the canopy, appears to be the more important factor with regards to the establishment of seedlings. A consequence of the establishment of an adequate canopy cover is the formation of suitable microclimatic conditions, such as moisture, litter depth, etc.

3.4.4. Are Planted Restoration Study Sites Facilitating Regeneration?

Ecosystem function, in addition to ecosystem structure, must be returned to a site in order for ecological restoration to be judged successful. Evidence of regeneration in all planted restoration sites, except the 'youngest' (P1) indicated that functional

processes necessary to initiate site regeneration, such as seed dispersal, were occurring (Reay & Norton, 1999a). This is particularly evident through the appearance of species not present as part of the restoration plantings themselves (e.g. *Carpodetus serratus*, *Coprosma grandifolia*, and *Dacrycarpus dacrydioides*). Further, the regeneration of species present with in the plantings is indicative of the occurrence of pollination, another vital ecosystem process (Reay & Norton, 1999a).

The majority of regeneration seedlings present within the eight study sites in which seedlings were found were almost certainly the result of bird dispersal. This assumption was based on the fact that seeds of most species found growing as seedlings occurred in the form of a berry or drupe. Numerous examples were found, within planted restoration study sites particularly (see Table 3.20), of regenerating seedlings occurring which were not present as part of the canopy stratum within that study site. Regenerating seedlings of woody species were found within five of the six planted restoration study sites. No evidence of regeneration was apparent within the P1 study site, which was the youngest study site with only two years lapse since planting took place. Age of the study site may not have been the sole cause of the lack of regeneration; rather the absence of appropriate microclimatic conditions to allow for germination and establishment may have been more influential. Distance to source sites does not appear to be a problem, due to its close proximity to the R3 study site.

Germination of a seed requires its survival within the physical environment and influences of surrounding plants (Simpson, 1992). Regeneration was not witnessed within the dense grass sward in planted restoration study sites. The poor competitive ability of seedlings within this environment is owed primarily to the fibrous root system, in addition to the physical smothering of seedlings by the dense grass sward (Reay & Norton, 1999a). Therefore, it was assumed that the presence of seedlings elsewhere within the planted restoration study sites is a strong indicator of the potential survival and growth of these seedlings. The formation of a canopy cover is a strong determining factor for the emergence and establishment of seedlings through the provision of suitable abiotic and microhabitat conditions.

The level of attractiveness of planted restoration study sites to bird species has been suggested to increase the dispersal of propagules into the site (Reay & Norton, 1999a). The majority of woody species present in the study area are suitable for bird dispersal; therefore it is apparent that birds play an important role in the dissemination of seeds throughout the study area. The role of birds as dispersal agents is discussed in more depth in Section 5.4.2.

3.4.5. Does Growth of Woody Species within Planted Restoration Study Sites Differ with Time Since Planting?

The results obtained specific to this section must be interpreted with a degree of caution due to the approximation of the height of species at the time of planting. However, general observations regarding height increases may be inferred from the resulting data. Substantial height increases of planted species have occurred since planting. Even in the 'youngest' study site investigated (P1), an estimated mean 260% increase in height since planting occurred. It was observed that the older the plantings, the greater the increase in mean height, with the exception of the P6 study site, when investigating the growth of 'all woody species' within. This may be due to the variety of growth forms of species contained within, rather than unsuitable conditions. For example, *Cordyline australis* has a distinctly 'lollipop' growth form compared with *Hebe elliptica*, which has a rounded form. Further, post-planting maintenance may not have occurred with sufficient vigour to enhance the initial establishment of species.

3.5. Summary

The results of the assessment of vegetation composition within the study area suggested that planted restoration sites are progressing, while facilitating the entry of secondary succession species. This implies that the initial stages necessary for a progression to a self-sustaining ecosystem are being provided for and enhanced. The current limiting factor to progression within the planted restoration study sites appears to be the lack of complete canopy cover.

4. GROUND ACTIVE INVERTEBRATES

4.1. Introduction

Ecological restoration studies frequently fail to provide sufficient consideration to the full range of species necessary to a self-sustaining, fully functioning ecosystem (Keesing & Wratten, 1998). Plant communities are generally given primary consideration, with faunal communities receiving less attention (Keesing & Wratten, 1998; Simmonds et al, 1994). Thus, the importance of fauna, particularly invertebrates, in the restoration process can be underestimated. Ground active invertebrates contribute greatly to the overarching goal of Holcim's restoration effort as they play a critical role in restoring structure, diversity and functioning to disturbed ecosystems (Majer & Nichols, 1998; Simmonds et al, 1994).

Invertebrates are ubiquitous and abundant, forming the majority of native fauna in New Zealand, with many species still be discovered and described. They perform a wide range of ecological functions and processes and are vulnerable to the loss of vegetation (Crisp et al, 1998; Simmonds et al, 1994). Invertebrates are sensitive to environmental variation (Williams, 1993), with their communities responding rapidly to subtle changes in their environment such as fine-scale variation in habitat structure (Golden & Crist, 2000).

Invertebrate communities therefore, offer a valuable tool, which is potentially more informative than monitoring physical or chemical variables, for interpreting and monitoring ecosystem change (Hutcheson, 1990). Because they are able to provide a more sensitive indication of the overall state of the ecosystem than plants, invertebrate communities are recognised as having much potential as habitat predictors (Oliver & Beattie, 1996) and as environmental indicators for measuring restoration success (Andersen & Sparling, 1997; Andersen, 1993; Williams, 1993). In Australia invertebrates, particularly ants, are used widely as indicators of ecosystem change (Abensperg-Traun & Steven, 1995) and in assessments of restoration success (Andersen & Sparling, 1997; Andersen, 1993). It must be recognised however, that

there is still a limited knowledge of the relationship between invertebrate communities and their habitat, especially in New Zealand (Hutcheson & Kimberley, 1999).

The purpose of this chapter is to (1) assess the success of the restoration plantings in re-establishing ground active invertebrate communities, and (2) investigate whether ground active invertebrates are good indicators of restoration success. These objectives will be achieved through the revelation of information regarding the diversity, abundance and distribution of ground active invertebrate communities in the nine study sites. The particular focus is on whether:

1. species composition of ground active invertebrates varied between planted restoration and remnant study sites;
2. species composition of ground active invertebrates varied between the six planted restoration study sites;
3. distribution of ground active invertebrates between the study sites was related to environmental variables; and whether
4. ground active invertebrates have potential to be used as environmental indicators for restoration success.

4.2. Methods

4.2.1. Field Methods

Measurement of the activity-abundance of ground active invertebrates was undertaken within 10 × 10 m study plots. Three study plots were located within each of the nine study sites, with the exception of the P6 study site, for which only two study plots were used, due to its limited size. Within each of the twenty-six study plots, five pitfall traps were established. Pitfall traps have been used widely as a method for sampling ground invertebrates (Crisp et al, 1998). The pitfall traps were left for six weeks over the December / January period (2001/2002), when invertebrate activity is greatest, and is most characteristic of a site (Hutcheson & Kimberley, 1999; Hutcheson, 1990). Pitfall traps were established 1 m in from each corner of the 10 × 10 m study plots, with a further pitfall trap placed in the centre of the plots (Figure 4.1).

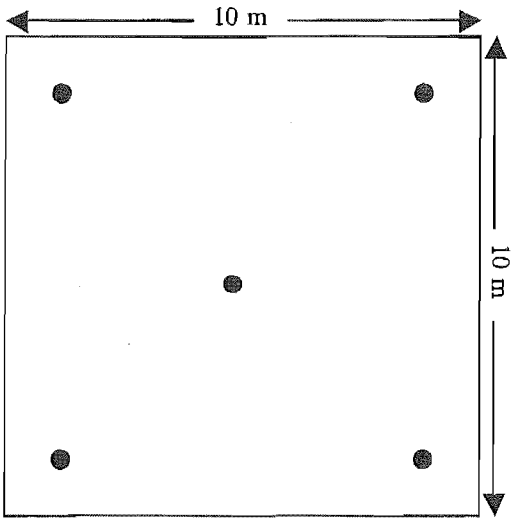


Figure 4.1: Location of pitfall traps within each of the twenty six 10 × 10 m study plots.

Pitfall traps were established by digging a small hole in the ground, with care taken to ensure as little disturbance took place as possible. The pitfall traps consisted of a plastic cup (70 mm diameter) placed into each hole, with its top level with the ground surface. At the beginning of the trap period 50 ml of 70% ethanol was poured into each cup, to act as a preservative (King et al, 1998). This preservative was topped up once, where necessary, during the six-week trapping period. A disadvantage to using such a preservative is the possibility that the alcohol may act as an attractant or deterrent to some invertebrate species (Simmonds et al, 1994). However, use of a preservative was essential to prevent decay of samples, and is used widely in pitfall trap studies. Because 70% ethanol was used in all pitfall traps in this study, the results are comparable between sites. A protective cover was placed over the top of each cup, held in place using wire supports, to prevent litter falling in or birds removing any of the samples, while still allowing invertebrates to enter the trap (Figure 4.2). As an additional deterrent to birds likely to damage the experimental set up, primarily *Gallirallus australis* (weka), a cage made from wire mesh (13 mm mesh size) was placed over the top of the pitfall traps and held in place using wire pegs.

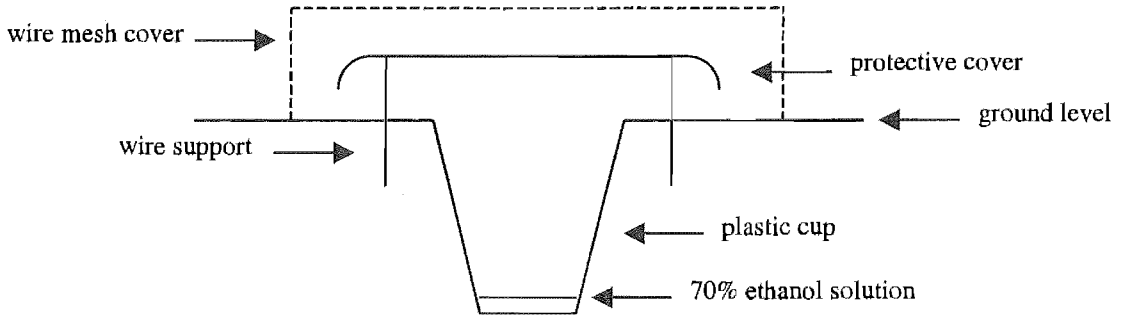


Figure 4.2: Set-up of pitfall traps.

4.2.1.1. Recognisable Taxonomic Units

Upon collection of the samples, all invertebrates found were sorted based on external morphology to recognisable taxonomic units (RTUs) and the frequency of those units was recorded (Hutcheson, 1990). Sorting invertebrates into distinguishable, RTUs, or morphospecies, as described by Oliver & Beattie (1993), enables sorting to be carried out by people who have received a relatively minimal amount of training, making the approach more accessible than if professionally trained taxonomists were required. This approach, however, can lead to underestimates or overestimates of the true diversity of invertebrates in a sample (Majer et al, 2002). For ease of speech, RTUs are hereafter referred to as species. A single example of each species identified was kept for the duration of this study as a reference collection.

4.2.2. Data Analysis

For the purposes of data analysis, invertebrate data was sorted into three categories:

1. all invertebrates,
2. Coleoptera (beetles), and
3. Araneida (spiders).

Analysis was repeated for each of the three categories.

4.2.2.1. Diversity Assessments

Diversity indices listed below were calculated for all three categories in order to elucidate potential patterns in species composition and abundance of ground invertebrates within the study area. However, the analysis of summed abundance classes was only applied to Coleoptera. For a more detailed account of the

methodology used for diversity indices see Section 3.2.2.2. Results from these indices are displayed in tabular form and graphically where appropriate (see Section 4.3.1.).

Species Richness

Species richness (*S*) was recorded as the total number of species observed within each study site. It is suggested that the number of species observed can be used for comparison of relative richness among study sites because observed species richness has been found to be highly correlated with true species richness (Oliver & Beattie, 1996).

Number of Individuals

The number of individual invertebrates collected within each study site was recorded.

Heterogeneity

Heterogeneity of invertebrates was determined using the Shannon index.

Dominance

Dominance within each of the nine study sites was calculated using the Berger-Parker index. The reciprocal form of this index was adopted so that an increase in the value of the index reflected an increase in diversity and reduction of dominance by individual species (Magurran, 1988).

Evenness

The modified Hill's ratio was adopted for this study to measure the equality of abundances of invertebrates within study sites (Alatalo, 1981). Maximum evenness (1.0) arises when all species are equally abundant; the least even study sites have an evenness value close to zero.

Summed Abundance Classes

Summed abundance classes (SAC) were used to analyse Coleoptera data as described by Hutcheson (1996) to provide a simple measure of diversity based on functional groups of invertebrates (Hutcheson & Kimberley, 1999). Coleoptera were categorised into functional groups (Hutcheson & Kimberley 1999; Hutcheson, 1996) at the family level: predators, herbivores (including all live plant feeders), and detritivores (including fungivores and scavengers) where appropriate, using the classification of

Harris & Burns (2000). Abundance classes were subsequently assigned to the Coleoptera using the defined levels used by Hutcheson & Kimberley (1999) of 0, 2, 5, 10, and 20+ specimens. These abundance classes were ascribed abundance class values of 1 to 5 respectively. The abundance class values were combined for each study site to provide a summed abundance score. This information is displayed graphically as the number of species, and the number of individuals, within each functional group (Section 4.3.1.2.).

Similarity

Jaccard's similarity coefficient was adopted throughout this study to determine similarity between study sites, based on presence / absence data.

4.2.2.2. Statistical Analysis

Diversity assessments for the nine study sites were compared statistically using one-way analysis of variance (ANOVA), with the aid of the statistical package SAS V8. Pairwise multiple comparisons were run using Tukey's test to determine the nature of differences detected by ANOVA. The level of significance set for all statistical testing was set at $\alpha = 0.05$.

4.2.2.3. Ordination

An ordination was undertaken using Detrended Correspondence Analysis (DCA) of invertebrate abundance data. DCA provides an indirect ordination of plot by species data, identifying the dominant species gradients independent of environmental factors. DCA was completed using CANOCO (ter Braak & Smilauer, 1998). The rationale behind using this indirect ordination method was to utilise it as a descriptive tool, investigating patterns within the distribution and abundance of invertebrate species over the nine study sites, and uncover underlying environmental gradients.

The axes of a DCA ordination are scaled in units of the average standard deviation of species turnover (SD) (Kent & Coker, 1992). A species appears, rises to its mode, and disappears within a span of approximately 4 SD. Likewise, a full turnover of species composition in samples occurs in 4 SD. A 50% change in species composition of a sample occurs within 1 SD or slightly more (Gauch, 1982). DCA scales axes in these SD units, thus axes can be of varying length, indicating the length of the community

gradient (Gauch, 1982). Eigenvalues are associated with each axis, representing the amount of variation explained by a particular axis.

Sample (study plot) and species ordination diagrams were computed for the twenty six study plots and seventy species of invertebrates found. Each point on the ordination graph corresponds to a particular study plot, or species. The distances between the sample points on the ordination are an approximation of the degree of similarity between study plots. For instance, if two study plots had identical species compositions then, they would occupy the same point on the ordination diagram. In a similar manner, species that have a similar distribution among study plots would occur close together in the species ordination. As species distributions deviate the distance between the points on the species ordination diagram increases (Kent & Coker, 1992).

The associations between selected environmental variables and the first and second axes of the sample ordination were assessed using non-parametric Spearman rank correlations. The critical value for the all correlations (df = 25) was $r = 0.337$ at a significance level of $\alpha = 0.05$. Kent & Coker (1992) note that significant linear relationships will only be found parallel to each axis using this approach.

4.3. Results

4.3.1. Diversity Assessments

4.3.1.1. All Invertebrates

Species Richness

The greatest mean number of distinct invertebrate species collected was 17.3, which was the mean number found in both P3 and P4 study sites (Figure 4.3a). P6 was found to have the fewest mean number of species of invertebrates ($S = 9.5$). The results of a one-way ANOVA for species richness of all invertebrate species by type of site (i.e. planted restoration or remnant sites) showed that no significant difference existed between the mean number of species collected in the two site types ($F = 0.35$, df = 1, $P = 0.561$). However, an ANOVA investigating mean species richness by different

planted restoration study sites suggested that a significant difference was present (F = 3.85, df = 5, P = 0.029). Multiple comparison tests undertaken using the Tukey's test did not expose the nature of these differences, possibly due to the degree of variance around the mean values (Table 4.1). An alternative multiple comparison test was used to investigate the differences. The LSD test showed that study sites P1 and P6 both had significantly fewer invertebrate species on average than study sites P2, P3, P4 and P5.

Table 4.1: Species richness (S) of all invertebrate species detected in study sites. Part (a) looks at descriptive statistics for remnants and planted restoration sites, while (b) details values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	12.0	17.0	14.2^{A}	0.7
	P	17	8.0	20.0	15.1^{A}	1.0
(b)	P1	3	8.0	13.0	11.0^{A}	1.5
	P2	3	15.0	18.0	17.0^{A}	1.0
	P3	3	15.0	20.0	17.3^{A}	1.5
	P4	3	13.0	20.0	17.3^{A}	2.2
	P5	3	12.0	19.0	16.3^{A}	2.2
	P6	2	9.0	10.0	9.5^{A}	0.5

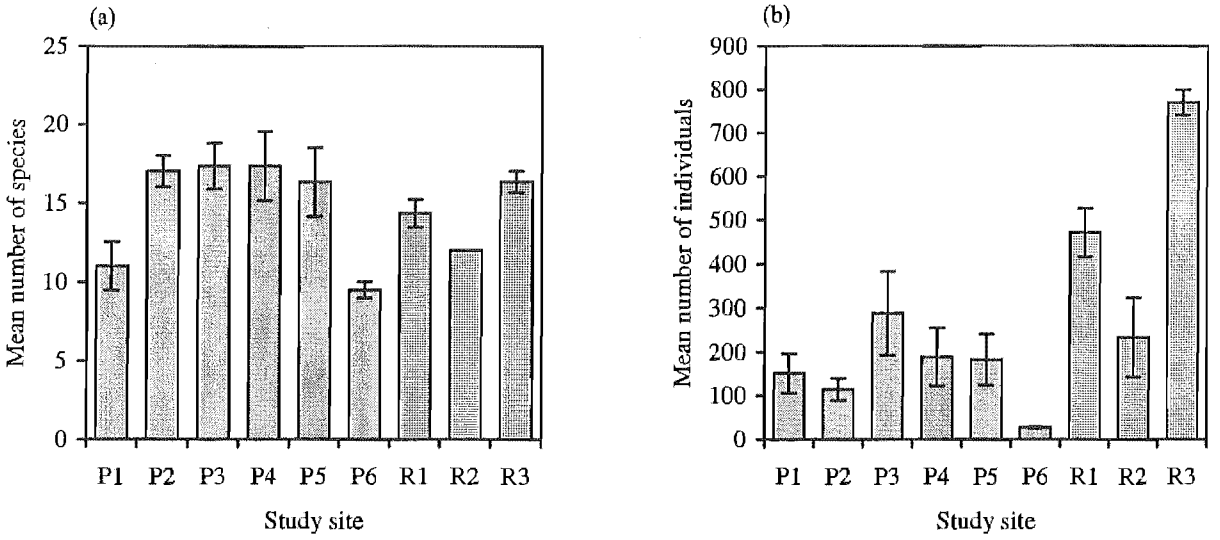


Figure 4.3: Graph (a) illustrates the total number of species (S) found in each study site; graph (b) shows the average number of individuals (N) collected for the category 'all invertebrates'. For both graphs, the error bars represent the standard error around the mean.

Number of Individuals

An ANOVA investigating differences in the mean number of individuals of invertebrate species collected in planted restoration sites and remnants indicated the presence of a significant difference (F = 20.78 df = 1, P = < 0.001). Further analysis using the Tukey's test revealed that remnant sites contained significantly more individuals of invertebrate species than found in planted restoration sites (Table 4.2). The greatest mean number of invertebrate individuals, for all study sites, was found in R3 (N = 770.0). The P3 study site had the highest mean number of invertebrate individuals found (N = 288.3) in planted restoration sites, while P6 had the lowest number of invertebrate individuals for all study sites (N = 27.0) (Figure 4.3b). No significant difference was detected in the mean number of individuals of invertebrates collected between the six planted restoration sites (F = 1.76, df = 5, P = 0.201).

Table 4.2: Number of invertebrate individuals collected in study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	127.0	828.0	491.7^{A}	83.9
	P	17	25.0	442.0	166.4^{B}	27.9
(b)	P1	3	67.0	222.0	151.3^{A}	45.3
	P2	3	86.0	165.0	114.7^{A}	25.3
	P3	3	114.0	442.0	288.3^{A}	95.3
	P4	3	56.0	265.0	188.7^{A}	66.6
	P5	3	94.0	293.0	182.0^{A}	58.6
	P6	2	25.0	29.0	27.0^{A}	2.0

Heterogeneity

P4 was the most heterogeneous study site, in terms of invertebrate species collected, (H' = 2.21). The least heterogeneous planted restoration site was P1 (H' = 1.12). R3 was the least heterogeneous study site overall (H' = 0.86) (Figure 4.4a). A significant difference was detected by an ANOVA for Shannon's diversity index by type of study site (F = 13.79, df = 1, P = 0.001). Planted restoration sites were found to be significantly more heterogeneous than remnant sites, by the Tukey's test. Similarly, a significant difference in H' was indicated between the six planted restoration sites (F = 4.97, df = 5, P = 0.013). The results of the Tukey's test revealed that P2 and P4 had significantly higher average values of H' than P1 (Table 4.3).

Table 4.3: Values of the Shannon index (*H'*) for 'all invertebrate' species collected within study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.79	1.95	1.17^{B}	0.13
	P	17	0.38	2.38	1.85^{A}	0.11
(b)	P1	3	0.38	1.58	1.12^{B}	0.37
	P2	3	1.88	2.38	2.17^{A}	0.15
	P3	3	1.74	1.95	1.81^{AB}	0.07
	P4	3	2.14	2.26	2.21^{A}	0.04
	P5	3	1.72	1.89	1.83^{AB}	0.05
	P6	2	1.98	2.07	2.03^{AB}	0.05

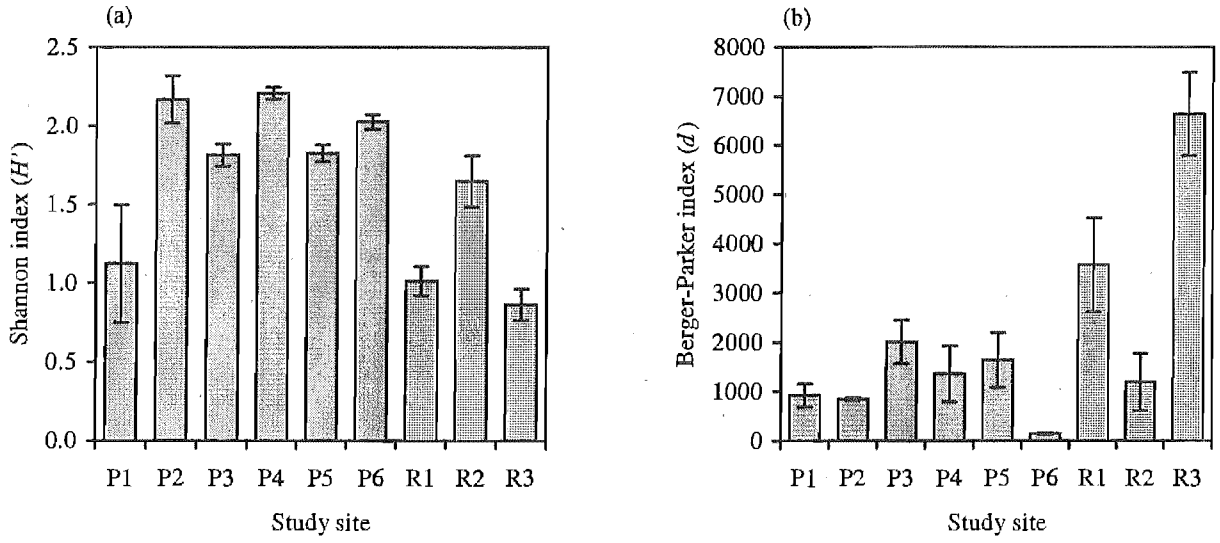


Figure 4.4: Graph (a) shows the results of the Shannon index (*H'*) for all invertebrate species collected in the nine study sites, while graph (b) illustrates the inverse Berger-Parker index (*d*) for all study sites. The error bars depict the standard error around the mean.

Dominance

The results of a one-way ANOVA for the inverse Berger-Parker dominance index (*d*) by study site type suggested that a significant difference occurred between remnant and planted restoration study sites (F = 14.04, df = 1, P = 0.001). The Tukey's test revealed that remnant sites had higher values for this dominance index than did planted restoration sites (Table 4.4). A further ANOVA was run to investigate potential differences in average *d* values between planted restoration sites. However, no significant difference was apparent (F= 2.25, df = 5, P = 0.121). R3 had the highest overall dominance value (*d* = 6633.84). P6 obtained the smallest dominance

value (d = 142.93), while P3 had a d value of 2007.68, the highest among planted restoration study sites (Figure 4.4b).

Table 4.4: Descriptive statistics of Berger-Parker dominance index (d) values obtained for invertebrate species collected in study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	557.06	8276.00	3796.30^{A}	885.26
	P	17	126.70	2670.58	1212.56^{B}	198.77
(b)	P1	3	508.96	1324.12	924.89^{A}	235.46
	P2	3	789.12	893.91	847.81^{A}	30.90
	P3	3	1170.67	2670.58	2007.68^{A}	441.65
	P4	3	304.29	2258.73	1357.42^{A}	569.28
	P5	3	619.80	2533.96	1638.08^{A}	555.95
	P6	2	126.70	159.17	142.93^{A}	16.23

Evenness

A slight significant difference in evenness values between planted restoration and remnant sites was detected by ANOVA (F = 4.68, df = 1, P = 0.041). The Tukey's test showed that planted restoration sites had a more even distribution of invertebrate species than did remnant study sites. A significant difference was suggested for evenness values between planted restoration study sites (F = 4.58, df = 5, P = 0.017). According to the Tukey's test, P6 had a significantly higher mean value for evenness than P1 (Table 4.5). P6 obtained the highest evenness value for all study sites (E = 0.82), while the lowest value over all study sites was found in R3 (E = 0.40). The least even distribution of invertebrate species within planted restoration sites was found in P1 (E = 0.46) (Figure 4.5).

Table 4.5: Descriptive statistics of values obtained for the modified Hill's ratio for evenness (E) of invertebrate species collected in study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.38	0.74	0.51^{B}	0.04
	P	17	0.34	0.82	0.62^{A}	0.03
(b)	P1	3	0.34	0.53	0.46^{B}	0.06
	P2	3	0.51	0.70	0.60^{AB}	0.05
	P3	3	0.58	0.73	0.66^{AB}	0.04
	P4	3	0.58	0.70	0.64^{AB}	0.03
	P5	3	0.54	0.73	0.61^{AB}	0.06
	P6	2	0.82	0.82	0.82^{A}	0.00

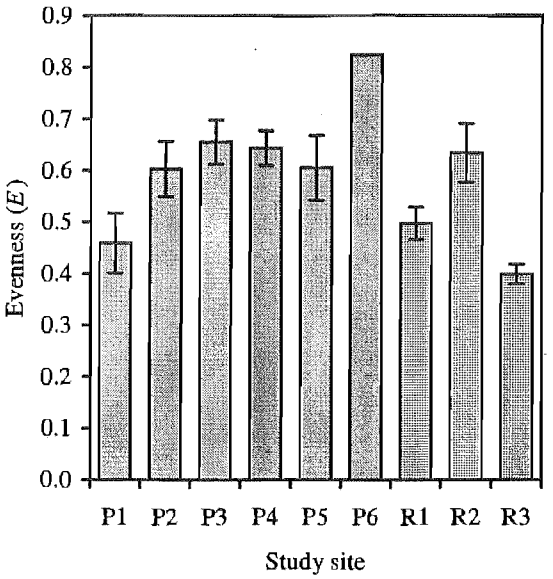


Figure 4.5: Results of the modified Hill's ratio of evenness (E) for 'all invertebrates' collected over each of the nine study sites. The error bars illustrate the standard error around the mean.

Similarity

An overall similarity coefficient of invertebrate species collected within remnant and planted restoration sites of $C_J = 0.48$ was found. The highest level of similarity in invertebrate species collected in study sites was between P4 and R2 ($C_J = 0.56$). The least similar study sites were P1 and P6, with a similarity coefficient of $C_J = 0.21$ (Table 4.6).

Table 4.6: Jaccard's similarity coefficients for similarity of invertebrate species between study sites.

	P1	P2	P3	P4	P5	P6	R1	R2	R3
P1	1.00								
P2	0.31	1.00							
P3	0.30	0.53	1.00						
P4	0.28	0.43	0.53	1.00					
P5	0.25	0.40	0.36	0.43	1.00				
P6	0.21	0.32	0.44	0.32	0.25	1.00			
R1	0.22	0.34	0.44	0.34	0.25	0.33	1.00		
R2	0.24	0.24	0.37	0.56	0.28	0.24	0.48	1.00	
R3	0.27	0.35	0.40	0.35	0.32	0.24	0.43	0.39	1.00

4.3.1.2. Coleoptera

Species Richness

No significant difference was found as the result of a one-way ANOVA for species richness of Coleoptera by type of study site (planted restoration or remnant sites) (F = 1.02, df = 1, P = 0.323). An ANOVA suggested differences in species richness of Coleoptera were detected in planted restoration sites (F = 8.83, df = 5, P = < 0.001). The Tukey's test revealed that P2 had significantly higher species richness than P1, P5, and P6. In addition to having significantly lower average species richness than P2, the average species richness for P1 was also found to be significantly less than that for P4 and P3 (Table 4.7). P2 had the greatest mean number of Coleoptera species collected (S = 6.7), while P1 had the fewest Coleoptera species collected overall with a mean of just 2.3 species caught in pitfall traps (Figure 4.6a).

Table 4.7: Species richness values (S) for Coleoptera located in study sites. Part (a) shows descriptive statistics or remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	3.0	8.0	5.2^{A}	0.5
	P	17	2.0	8.0	4.6^{A}	0.4
(b)	P1	3	2.0	3.0	2.3^{CD}	0.3
	P2	3	6.0	8.0	6.7^{A}	0.7
	P3	3	5.0	5.0	5.0^{AB}	0.0
	P4	3	5.0	6.0	5.3^{AB}	0.3
	P5	3	3.0	5.0	4.0^{BC}	0.6
	P6	2	3.0	5.0	4.0^{BC}	1.0

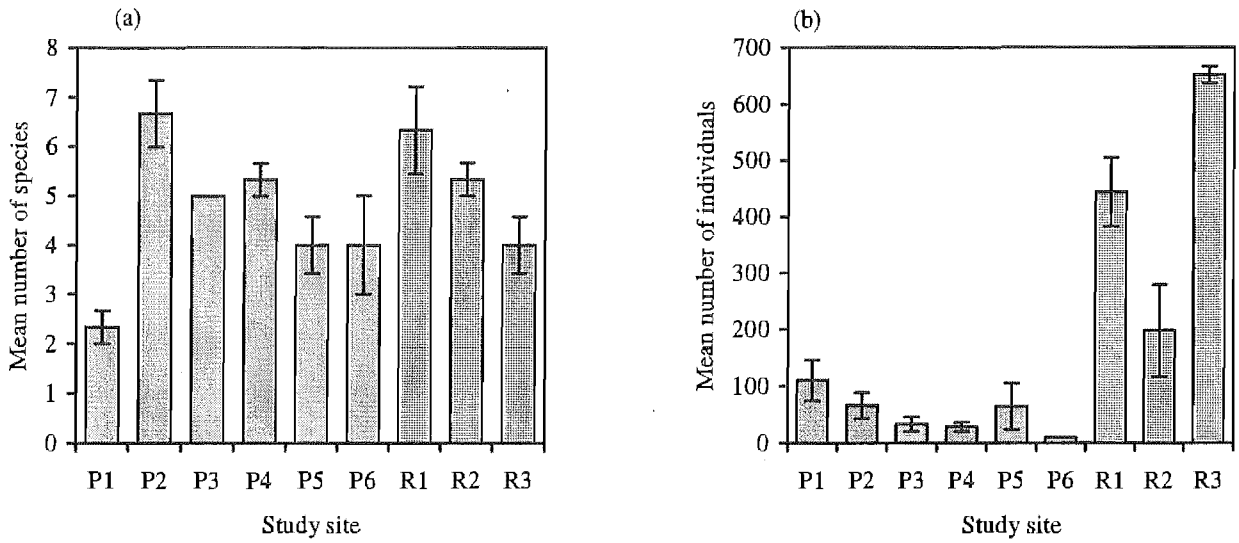


Figure 4.6: Graph (a) illustrates the mean number of different Coleoptera species, while graph (b) represents the mean number of Coleoptera individuals collected in each study site. The error bars depict the standard error around the mean values, for each study site.

Number of Individuals

The largest numbers of Coleoptera were found in R3, with an average of 651.7 Coleoptera collected in study plots. P6 had the fewest Coleoptera collected, with a mean of 9.5 individuals trapped (Table 4.8). An average of 110.3 individuals of Coleoptera was found in study plots within P1, the most for planted restoration study sites (Figure 4.6b).

Table 4.8: Number of individuals of Coleoptera (*N*) collected in study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	111.0	680.0	431.2^{A}	72.0
	P	17	9.0	158.0	54.5^{B}	12.1
(b)	P1	3	40.0	158.0	110.3^{A}	35.9
	P2	3	31.0	109.0	66.0^{A}	22.9
	P3	3	9.0	53.0	33.3^{A}	12.9
	P4	3	16.0	43.0	28.3^{A}	7.9
	P5	3	14.0	146.0	64.3^{A}	41.2
	P6	2	9.0	10.0	9.5^{A}	0.5

The results of a one-way ANOVA for the number of individuals of Coleoptera collected by site type indicated that a statistically significant difference existed between the number of Coleoptera found in planted restoration sites and remnants

(F = 48.57, df = 1, P = < 0.001). The Tukey's test showed that significantly more Coleoptera were collected in remnant sites than in planted restoration sites. No significant difference was apparent in the mean number of Coleoptera collected in the various planted restoration sites (F = 1.72, df = 5, P = 0.212).

Heterogeneity

The mean value of H' was not found to differ significantly between planted restoration sites and remnants (F = 0.60, df = 1, P = 0.448). ANOVA did reveal an apparent significant difference in H' values between planted restoration sites (F = 7.36, df = 5, P = 0.003). According to the Tukey's test, P1 had significantly lower values of H' than P4 and P2. P4 was also found to have significantly greater H' values than P5 (Table 4.9). The least heterogeneous study site, in terms of Coleoptera species collected, was P1 (H' = 0.20); while P4 was the most heterogeneous study site (H' = 1.35) (Figure 4.7a).

Table 4.9: Values of the Shannon Index (H') for Coleoptera species collected throughout study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.07	1.38	0.71^{A}	0.15
	P	17	0.14	1.52	0.86^{A}	0.12
(b)	P1	3	0.14	0.27	0.20^{B}	0.04
	P2	3	0.95	1.44	1.25^{A}	0.15
	P3	3	0.41	1.52	0.94^{AB}	0.32
	P4	3	1.34	1.37	1.35^{AC}	0.01
	P5	3	0.27	0.54	0.44^{BC}	0.08
	P6	2	0.80	1.30	1.05^{AB}	0.25

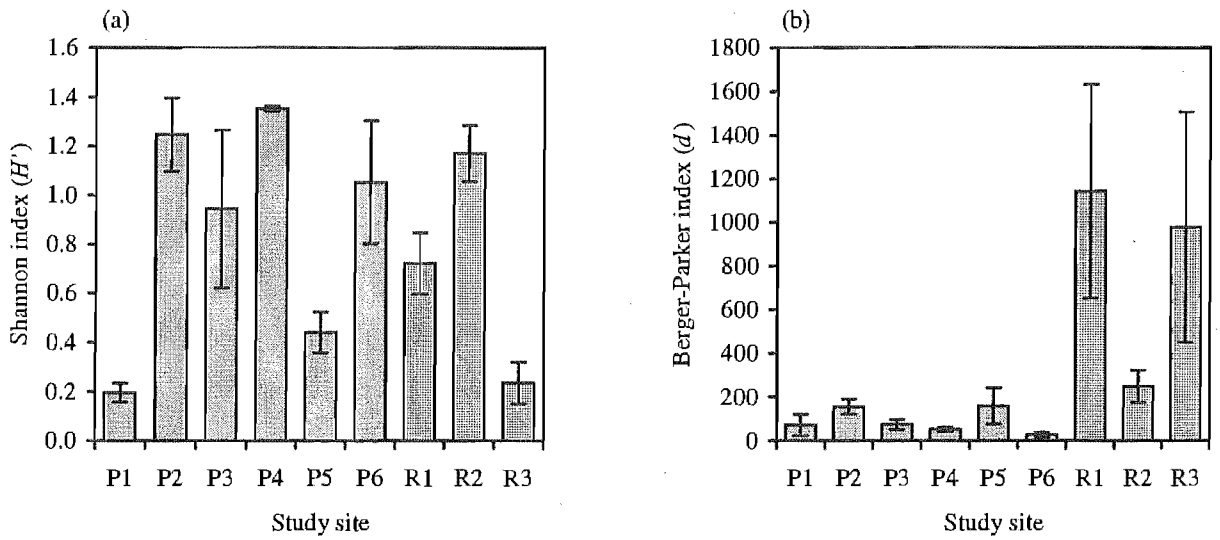


Figure 4.7: Graph (a) illustrates the results of the Shannon diversity index for Coleoptera, while graph (b) shows the inverse Berger-Parker index for all study sites. The error bars illustrate the standard error around the mean.

Dominance

The results of ANOVA suggested that a significant difference occurred in the value of *d* between planted restoration sites and remnants ($F = 14.77$, df = 1, $P = < 0.001$). The Tukey's test showed that remnant sites had a mean *d* value significantly greater than that for planted restoration sites (Table 4.10). No significant difference was found from an ANOVA investigating differences between *d* values in planted restoration sites ($F = 1.35$, df = 5, $P = 0.313$).

Table 4.10: Dominance values (*d*) for Coleoptera species collected in study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	59.36	2114.34	788.58^{A}	250.28
	P	17	14.41	313.92	93.64^{B}	19.88
(b)	P1	3	14.41	167.29	71.45^{A}	48.21
	P2	3	90.80	208.25	155.92^{A}	34.52
	P3	3	30.00	107.10	74.02^{A}	22.92
	P4	3	39.33	71.07	54.44^{A}	9.57
	P5	3	29.17	313.92	158.74^{A}	83.19
	P6	2	16.43	37.80	27.11^{A}	10.69

R1 was found to have a high dominance value for Coleoptera species (d = 1141.85), which reflects the lack of dominance of a few species within the study site (Figure 4.7b). P6 had a very low dominance value (d = 27.11) suggesting that only a few species dominated the numbers of Coleoptera collected within the study site. The highest dominance value within planted restoration sites was recorded for P5 (d = 158.74).

Evenness

No statistically significant difference was evident according to an ANOVA between evenness values in planted restoration and remnant sites (F = 0.00, df = 1, P = 0.969). Evenness values for the six planted restoration sites did not vary significantly (F = 2.19, df = 5, P = 0.130). P4 had the highest value for Hill's modified ratio for evenness (E = 0.72) in planted restoration sites, surpassed only by R2 (E = 0.75) (Table 4.11). P1 and P5 shared the lowest evenness value for planted restoration sites of 0.45. R3 had the least even selection of Coleoptera species with E = 0.41 (Figure 4.8).

Table 4.11: Values for the modified Hill's ratio for evenness (E) of Coleoptera species collected throughout study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.28	0.84	0.59^{A}	0.06
	P	17	0.40	0.91	0.58^{A}	0.04
(b)	P1	3	0.40	0.53	0.45^{A}	0.04
	P2	3	0.53	0.69	0.61^{A}	0.05
	P3	3	0.42	0.91	0.62^{A}	0.15
	P4	3	0.60	0.83	0.72^{A}	0.07
	P5	3	0.41	0.52	0.45^{A}	0.03
	P6	2	0.67	0.69	0.68^{A}	0.01

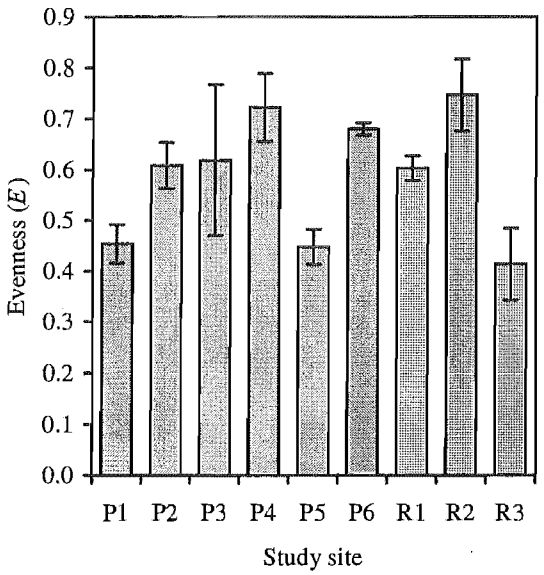


Figure 4.8: Results of the modified Hill's ratio of evenness for Coleoptera species collected in each of the nine study sites. The error bars depict the standard error around the mean.

Similarity

An overall similarity of Coleoptera species between remnant and planted restoration sites of $C_J = 0.56$ was found. The most similar study sites, in terms of presence of Coleoptera species were P2 and P3 ($C_J = 0.80$). P1 and R2 were the least similar with a Jaccard's coefficient of just 0.08 (Table 4.12).

Table 4.12: Jaccard's similarity coefficients for similarity of Coleoptera species between study sites.

	P1	P2	P3	P4	P5	P6	R1	R2	R3
P1	1.00								
P2	0.36	1.00							
P3	0.30	0.80	1.00						
P4	0.30	0.67	0.64	1.00					
P5	0.17	0.46	0.42	046	1.00				
P6	0.25	0.50	0.63	0.36	0.27	1.00			
R1	0.27	0.46	0.55	0.46	0.20	0.56	1.00		
R2	0.08	0.29	0.33	0.38	0.21	0.18	0.55	1.00	
R3	0.09	0.42	0.50	0.55	0.33	0.50	0.60	0.36	1.00

Summed Abundance Classes

Predatory Coleoptera were more prevalent, in terms of number of different species, than herbivorous Coleoptera (Figure 4.9a). R1 and R2 contained more herbivorous Coleoptera species than predatory. In all three remnant sites considerably more individuals of herbivorous Coleoptera were found than predatory ones (Figure 4.9b). In planted restoration sites, predatory Coleoptera dominated in terms of species and individuals collected.

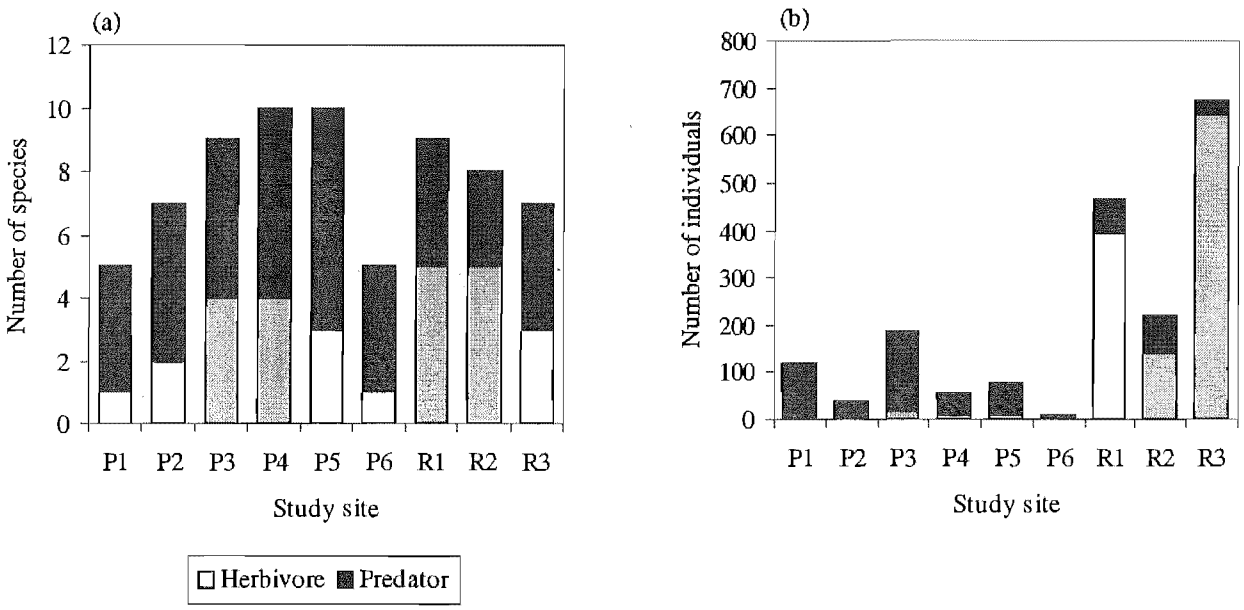


Figure 4.9: Summed abundance classes for Coleoptera species. Graph (a) shows the relationship between the number of species collected and functional group for each study site. Graph (b) illustrates the relationship between the number of individuals of Coleoptera and functional group for each study site. The key applies to both graphs and refers to Coleoptera functional groupings.

4.3.1.3. Araneida

Species Richness

The highest mean number of Araneida species were collected in P2 (S = 6.7). The lowest mean number of Araneida species were trapped in P6 (Table 4.13; Figure 4.10a). The results of a one-way ANOVA for species richness of Araneida species by study site type (remnant or planted restoration sites) found no significant difference between the two (F = 0.04, df = 1, P = 0.834). Neither was a significant difference detected in species richness of Araneida species between planted restoration sites (F = 0.38, df = 5, P = 0.854).

Table 4.13: Species richness values (*S*) for Araneida species collected in study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.0	6.0	3.0^{A}	0.6
	P	17	1.0	5.0	3.1^{A}	0.3
(b)	P1	3	1.0	5.0	2.7^{A}	1.2
	P2	3	2.0	5.0	3.7^{A}	0.9
	P3	3	3.0	4.0	3.3^{A}	0.3
	P4	3	3.0	4.0	3.3^{A}	0.3
	P5	3	3.0	3.0	3.0^{A}	0.0
	P6	2	2.0	3.0	2.5^{A}	0.5

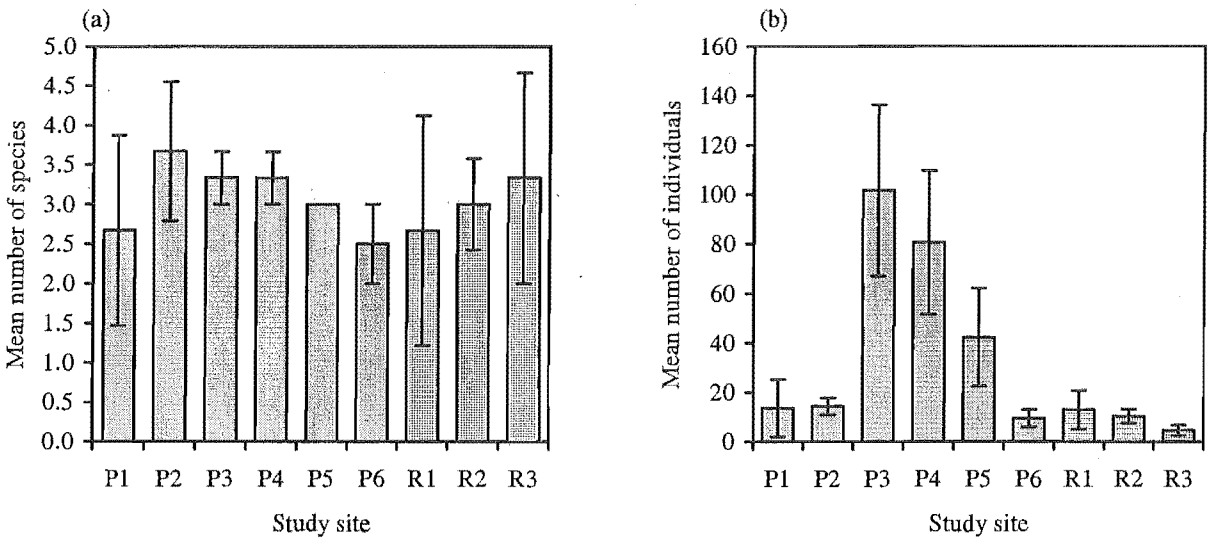


Figure 4.10: Graph (a) illustrates the number of Araneida species collected in each study site; graph (b) shows the average number of individuals collected per study site. The error bars illustrate the standard error around the mean.

Number of Individuals

Generally, more individuals of Araneida were found within planted restoration sites, than within remnant sites (Figure 4.10b). P3 had the greatest mean number of Araneida found (N = 101.7). P6, on average, had the least number of Araneida (N = 9.5), for planted restoration sites. R3 had the least number of Araneida species overall, with a mean of 4.7 recorded. A statistically significant difference between the mean number of Araneida collected in remnant and planted restoration sites was detected by ANOVA (F = 4.85, df = 1, P = 0.038). According to the Tukey's test, planted restoration sites had significantly more Araneida than remnant sites (Table 4.14).

No significant difference was evident between the mean number of Araneida collected in different planted restoration sites ($F = 3.14$, df = 5, $P = 0.053$).

Table 4.14: Number of individuals of Araneida (N) collected within the nine study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.0	27.0	9.3^{B}	2.8
	P	17	2.0	143.0	45.5^{A}	11.7
(b)	P1	3	2.0	37.0	13.7^{A}	11.7
	P2	3	8.0	20.0	14.3^{A}	3.5
	P3	3	33.0	143.0	101.7^{A}	34.6
	P4	3	23.0	112.0	80.7^{A}	28.9
	P5	3	11.0	79.0	42.3^{A}	19.8
	P6	2	4.0	1.0	9.5^{A}	3.5

Heterogeneity

Shannon index values did not vary significantly between remnants and planted restoration sites ($F = 0.76$, df = 1, $P = 0.393$); nor were statistically significant differences in heterogeneity detected between the six planted restoration sites ($F = 1.17$, df = 5, $P = 0.382$) (Table 4.15). P2 had the greatest heterogeneity, in terms of Araneida species, ($H' = 0.89$), while P1 had the lowest ($H' = 0.48$) (Figure 4.11a).

Table 4.15: Values of the Shannon index (H') for Araneida species collected throughout study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.00	1.73	0.76^{A}	0.17
	P	17	0.00	1.09	0.63^{A}	0.07
(b)	P1	3	0.00	0.75	0.48^{A}	0.24
	P2	3	0.56	1.09	0.89^{A}	0.17
	P3	3	0.22	0.78	0.43^{A}	0.18
	P4	3	0.53	0.74	0.66^{A}	0.07
	P5	3	0.60	0.64	0.62^{A}	0.01
	P6	2	0.56	0.86	0.71^{A}	0.15

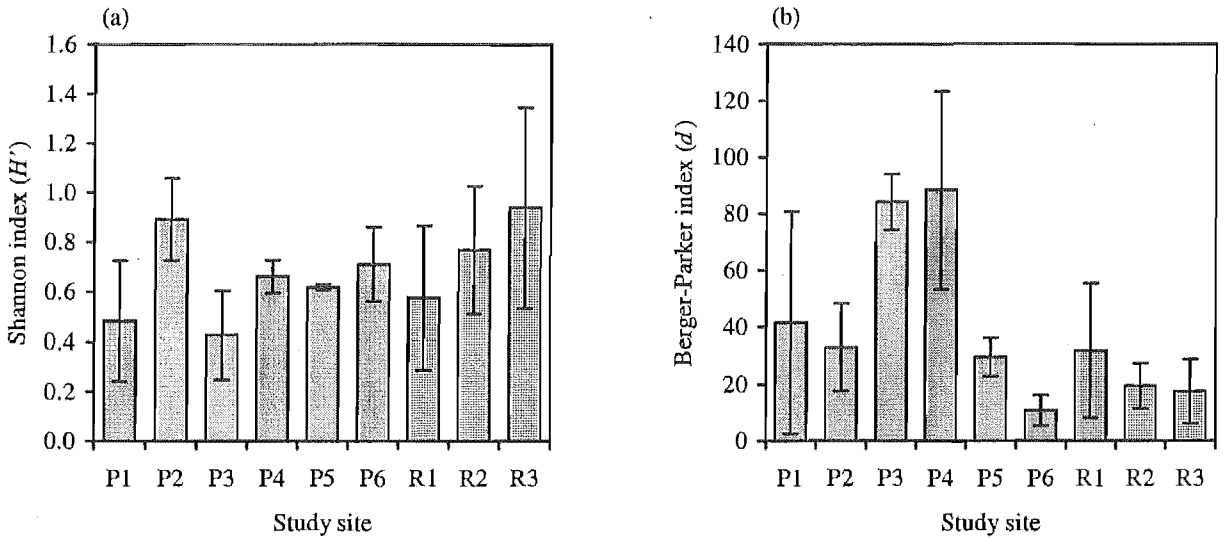


Figure 4.11: Graph (a) illustrates results of the Shannon index (H') for Araneida species, while graph (b) shows the Berger-Parker index (d). For both graphs, the error bars depict the standard error around the mean.

Dominance

Dominance values did not differ significantly between planted restoration sites or remnants (F = 2.82, df = 1, P = 0.106); nor were significant differences detected between the six planted restoration sites (F = 1.56, df = 5, P = 0.251) (Table 4.16). P4 had highest value for dominance for all study sites (d = 88.33). P6 was found to have the lowest dominance value in comparison with the other eight study sites (d = 10.79) (Figure 4.11b).

Table 4.16: Dominance values (d) for Araneida species collected throughout study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.00	77.85	22.89^{A}	8.21
	P	17	1.00	127.82	50.06^{A}	10.87
(b)	P1	3	1.00	119.68	41.56^{A}	39.07
	P2	3	5.33	58.21	32.99^{A}	15.31
	P3	3	67.06	101.15	84.08^{A}	9.84
	P4	3	18.60	127.82	88.33^{A}	34.97
	P5	3	22.40	42.95	29.52^{A}	6.72
	P6	2	5.33	16.24	10.79^{A}	5.45

Evenness

Values of the modified Hill's ratio for evenness did not differ significantly between planted restoration sites and remnants ($F = 0.31$, df = 1, $P = 0.583$); nor did significant differences occur between the six planted restoration sites ($F = 0.67$, df = 5, $P = 0.653$) (Table 4.17). The extremes for evenness values were found within remnant sites. R3 had the highest value (E = 0.84), while R1 was the least even study site, in terms of the number of Araneida species collected (E = 0.42). Within planted restoration sites P6 was the most even site (E = 0.78), and P3 was the least even study site (E = 0.48) (Figure 4.12).

Table 4.17: Modified Hill's ratio for evenness values (*E*) for Araneida collected throughout study sites. Part (a) shows descriptive statistics for remnants and planted restoration sites, while (b) lists values for the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.00	0.96	0.66^{A}	0.10
	P	17	0.00	1.00	0.61^{A}	0.05
(b)	P1	3	0.00	1.00	0.50^{A}	0.29
	P2	3	0.59	0.79	0.70^{A}	0.06
	P3	3	0.41	0.60	0.48^{A}	0.06
	P4	3	0.57	0.67	0.63^{A}	0.03
	P5	3	0.56	0.68	0.60^{A}	0.04
	P6	2	0.77	0.79	0.78^{A}	0.01

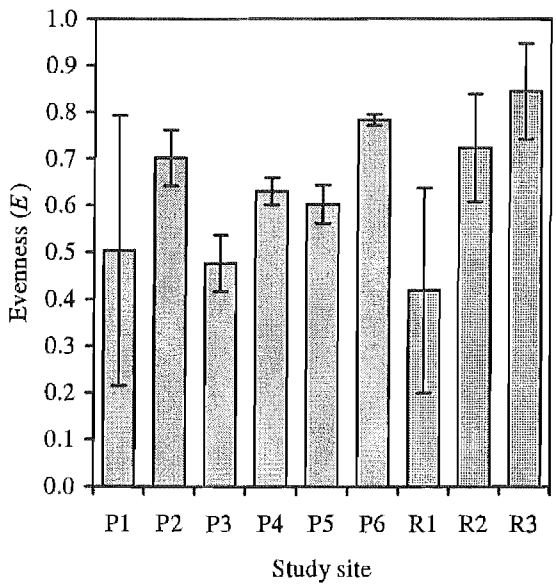


Figure 4.12: Results of the modified Hill's ratio of evenness for Araneida species collected in each of the nine study sites. The error bars depict the standard error around the mean.

Similarity

An overall similarity of Araneida species between remnant and planted restoration sites of $C_J = 0.50$ was found. P4 and R3 were the least similar study sites ($C_J = 0.10$) and P4 and P5 were found to share the most Araneida species in common ($C_J = 0.80$) (Table 4.18).

Table 4.18: Jaccard's similarity coefficients for similarity of Araneida species between study sites.

	P1	P2	P3	P4	P5	P6	R1	R2	R3
P1	1.00								
P2	0.25	1.00							
P3	0.50	0.57	1.00						
P4	0.13	0.38	0.25	1.00					
P5	0.14	0.25	0.29	0.80	1.00				
P6	0.33	0.29	0.33	0.14	0.17	1.00			
R1	0.33	0.25	0.29	0.13	0.14	0.17	1.00		
R2	0.33	0.43	0.29	0.13	0.14	0.17	0.60	1.00	
R3	0.25	0.33	0.38	0.10	0.11	0.50	0.67	0.43	1.00

4.3.2. Ordination

4.3.2.1. All Invertebrates

Figure 4.13 illustrates the study site ordination plot for the DCA, using all invertebrate data collected ($n = 26$ study plots). The gradient length of axis 1 was 4.735, and 2.324 for axis 2. Axis 1 had an eigenvalue of 0.816, explaining 20.0% of the total variation within the invertebrate data, while axis 2 accounted for 8.3% of the variation (axis 2 eigenvalue = 0.338).

Remnants and planted restoration study sites are clearly separated along axis 1. More than 4 SD separate P1 and all remnant study plots, indicating that there was a large turnover of invertebrate species between the aforementioned study plots. Remnant study plots are clumped together along both axes, indicative of similar species composition within all remnant study plots. A large degree of separation is present between planted restoration study plots along both axes. Planted restoration sites are separated by approximately 3 SD along both axis 1 and axis 2. From this, it can be inferred that there is more than 50% invertebrate species turnover between P1 and the study plots in P4 and P5 as they are separated by more than 1 SD. Additionally, more than 1 SD separated the study plots of P6 and P5, thereby inferring that these two study sites shared less than half their invertebrate species in common.

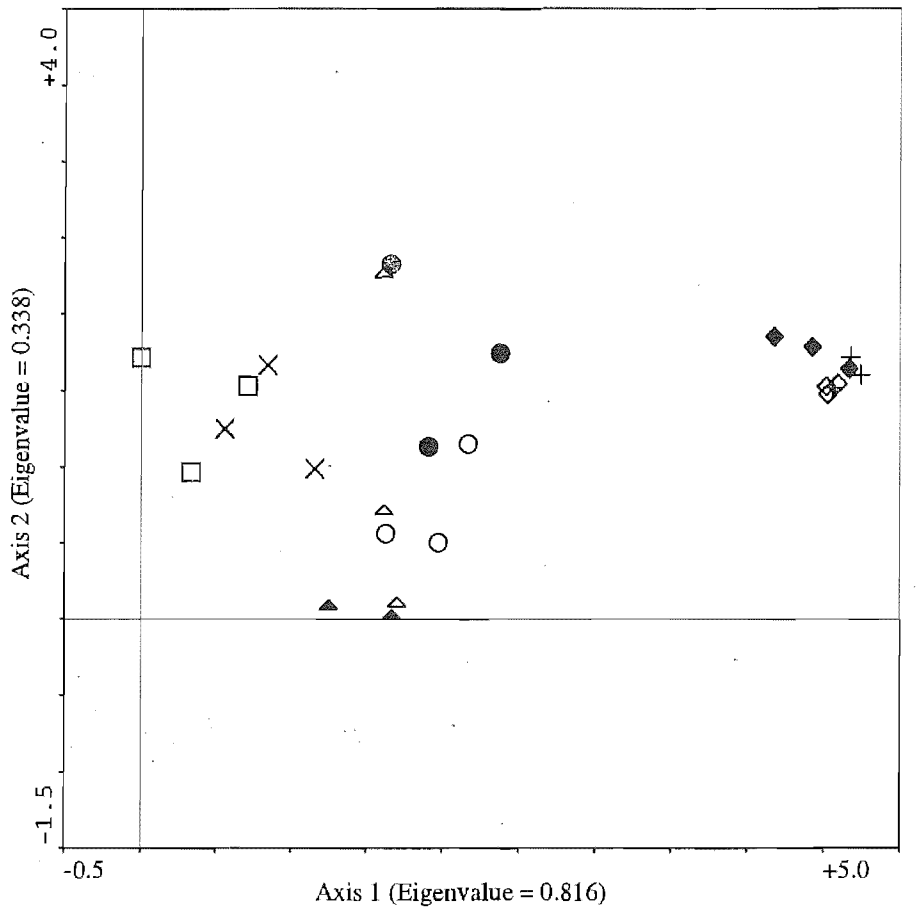


Figure 4.13: Ordination of 'all invertebrate' species by site using Detrended Correspondence Analysis. The symbols represent the nine different study sites: □ = P1, × = P2, Δ = P3, ○ = P4, ● = P5, ▲ = P6, + = R1, ♦ = R2, ◊ = R3.

Four environmental factors (litter depth, cover of woody species, dominance *d* of woody species, and *E* of woody species) were significantly correlated with the position of sites along either axis 1or 2 (Table 4.19). Axis 1 was very highly correlated with litter depth, inferring that, of the measured variables, litter depth was the principle factor influencing the position of study plots and species along this axis. Average cover of woody species appears to be a cause of separation along both axis 1 and axis 2, as it was significantly correlated with each. An additional cause of variation in position of study plots and species along axis 1 appears to be the dominance values of woody vegetation. The modified Hill's ratio for evenness was positively correlated with the position of study plots and species along axis 2; however, a significant negative correlation was present between evenness values of woody vegetation and the variation along axis 1.

Table 4.19: Spearman rank correlation between variables and position of study plots on axis 1 and 2 for the ordination of invertebrate data from all study plots (n = 26). The symbol * indicates a significant correlation at α = 0.05. The critical value for df = 25 was r = 0.337.

Environmental Variable	Axis 1	Axis 2
Aspect	- 0.272	- 0.120
Slope	0.229	0.084
Litter depth	0.878*	0.282
Species richness (vegetation)	0.239	- 0.99
Cover (vegetation)	0.533*	0.353*
H' (vegetation)	- 0.244	- 0.156
d (vegetation)	0.509*	0.235
E (vegetation)	- 0.379*	0.402*

An ordination diagram of invertebrate species (n = 70) collected over the nine study sites reflected a less clear demarcation between study sites (Figure 4.14). Species that appear in the middle of the ordination diagram are not indicative of any particular study site; as such they may appear in most or all study plots. Species present in the far right or left are more likely to have been found in remnant and planted restoration study sites respectively.

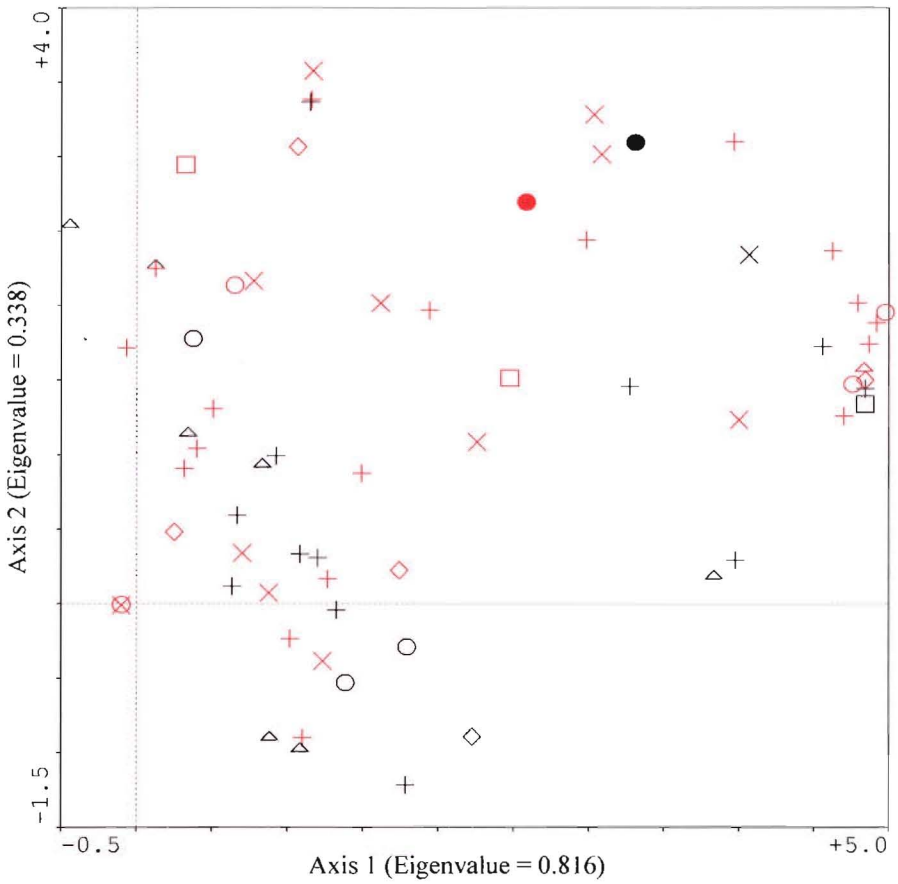


Figure 4.14: Ordination of 'all invertebrate' species collected over the nine study sites. The following symbols represent different orders of identified invertebrate species; + = Araneida, + = Coleoptera, × = Spirobolida, × = Diptera, ○ = Gastropoda, ○ = Hemiptera, Δ = Hymenoptera, Δ = Blattodea, ◊ = Isopoda, ◊ = Lepidoptera, □ = Acarina, □ = Amphipoda, ● = Oligochaeta, ● = Orthoptera.

A further DCA was run for the invertebrate data with rare species down-weighted (see Figure 4.15). Down weighting of rare species was done to determine whether rare species were disproportionately distorting the analysis. This data manipulation did not appear to significantly increase the amount of variation explained by either axis (22.1% by axis 1; 8.7% by axis 2) nor to alter the relative ordering of sites, resulting in the decision to run all further DCA using unmodified data.

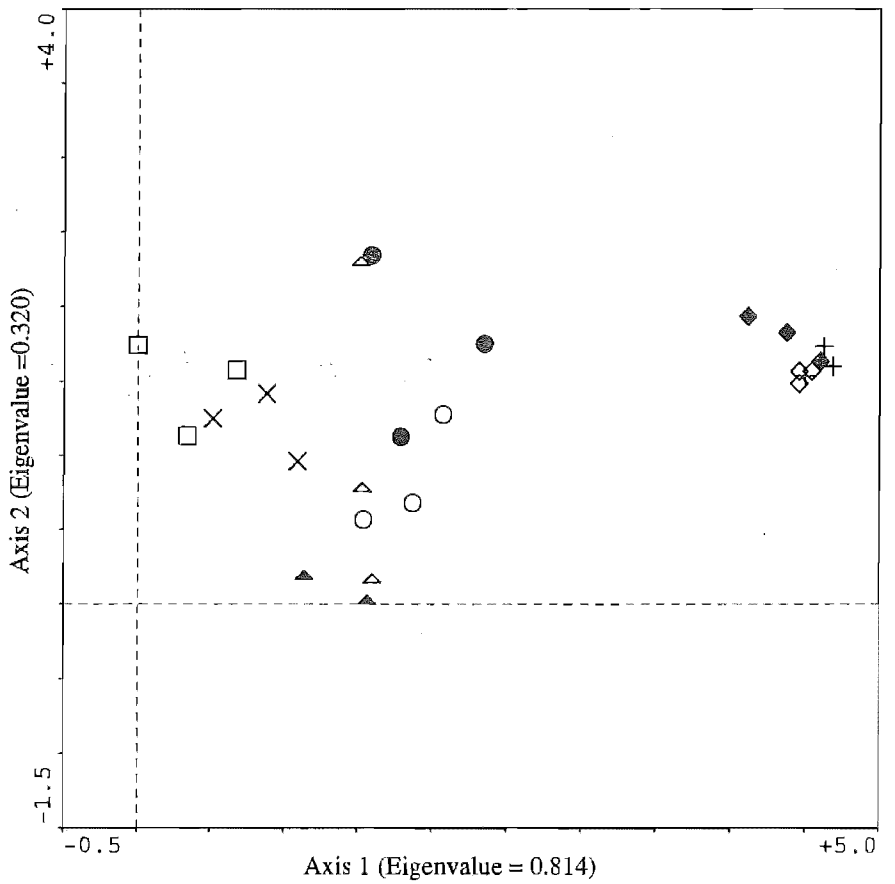


Figure 4.15: Ordination diagram of invertebrate species by site, with rare species down-weighted, using Detrended Correspondence Analysis. The symbols represent the nine different study sites: □ = P1, × = P2, Δ = P3, ○ = P4, ● = P5, ▲ = P6, + = R1, ♦ = R2, ◊ = R3.

4.3.2.2. Coleoptera

An ordination diagram for the DCA of Coleoptera species by study site (n = 26 plots) is shown in Figure 4.16. The gradient length of axis 1 was 5.194, and 2.084 for axis 2. Axis 1 had an eigenvalue of 0.918. 28.1% of variation within Coleoptera data was explained by axis 1. Axis 2 explained 10.7% of the variation, with an eigenvalue of 0.348.

Remnant and planted restoration study sites were clearly separated along axis 1. Study plots from all study sites covered more than 5 SD along axis 1. Thus it can be inferred that study sites P1 possibly shared no Coleoptera species in common with samples collected in any of the three remnant study sites. Study plots from all remnant sites overlapped on the ordination diagram, indicating that species and abundances of Coleoptera collected within the three study sites were highly similar. Study plots from planted restoration sites however, appeared more varied in their position on the ordination diagram. Over 2 SD separated the most divergent of the study plots. This suggests that many of the study plots shared fewer than 50% of Coleoptera species in common. However, some ubiquitous Coleoptera would have been detected over all planted restoration study sites.

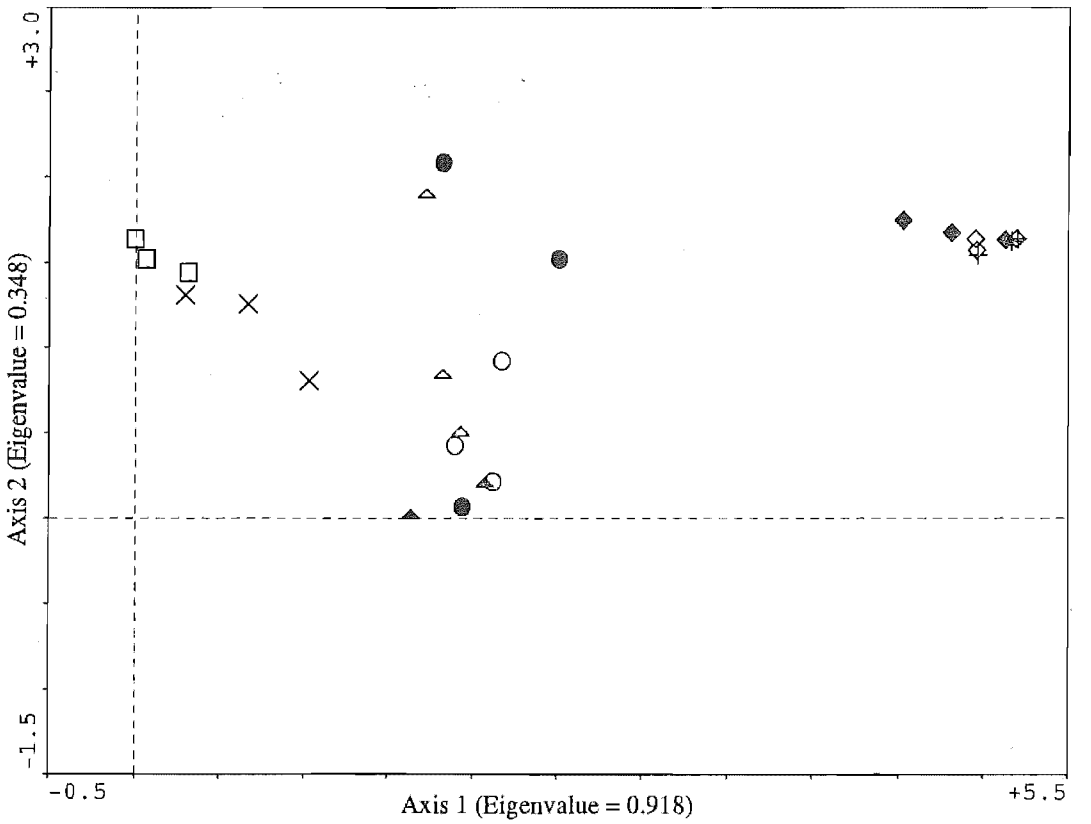


Figure 4.16: Ordination of Coleoptera species by site using Detrended Correspondence Analysis. The symbols represent the nine different study sites: □ = P1, × = P2, Δ = P3, ○ = P4, ● = P5, ▲ = P6, + = R1, ♦ = R2, ◊ = R3.

Litter depth was highly correlated with both axis 1 and axis 2, indicating that this environmental factor was detected to be a source of significant variation along both axes (Table 4.20). Cover, dominance, and evenness of woody vegetation were all found to be significantly correlated with axis 1. The latter environmental variable, evenness of woody vegetation, was negatively correlated with axis 1; indicating planted restoration sites had a more even distribution of woody species than remnant study sites.

Table 4.20: Spearman rank correlation between variables and position of study plots on axis 1 and axis 2 for the ordination of Coleoptera data from all study plots (n = 26). The symbol * indicates a significant correlation at $\alpha = 0.05$. The critical value for df = 25 was $r = 0.337$.

Environmental Variable	Axis 1	Axis 2
Aspect	- 0.325	- 0.228
Slope	0.205	0.054
Litter depth	0.882*	0.394*
Species richness (vegetation)	0.231	0.003
Cover (vegetation)	0.485*	0.312
H' (vegetation)	- 0.238	- 0.181
d (vegetation)	0.550*	0.274
E (vegetation)	- 0.396*	- 0.209

4.3.2.3. Araneida

Figure 4.17 illustrates an ordination diagram of Araneida species by site (n = 26). Axis 1 had a gradient length of 4.008 and the gradient length of axis 2 was 3.145. Axis 1 had an eigenvalue of 0.726, which accounted for 19.2% of the total variation within Araneida data. The eigenvalue of axis 2 (0.422) explained 11.2% of the variation.

Araneida species located within study sites were quite divergent in type and abundance. Study plots were spread over 4 SD, along axis 1, indicating that full species turnover took place between some of the selected study sites. More than 3 SD separated study plots on axis 2. The distinction between Araneida composition in remnant and planted restoration sites was less marked than that which was found for Coleoptera.

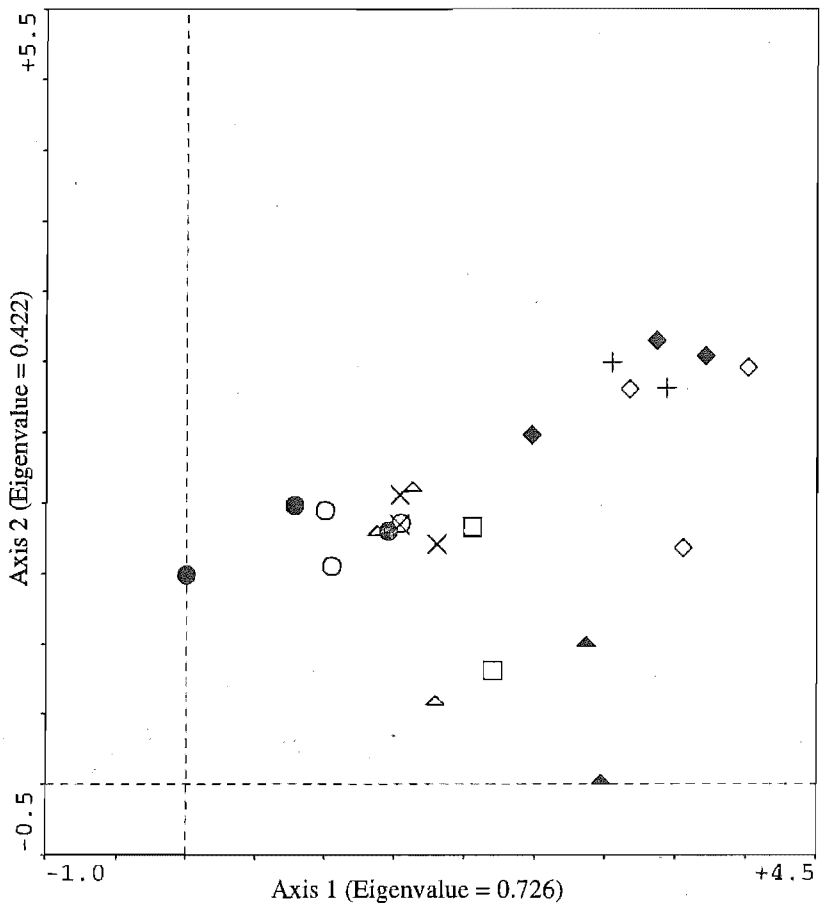


Figure 4.17: Ordination of Araneida species by site using Detrended Correspondence Analysis. The symbols represent the nine different study sites: □ = P1, × = P2, Δ = P3, ○ = P4, ● = P5, ▲ = P6, + = R1, ♦ = R2, ◊ = R3.

Numerous environmental variables were accountable for the position of study plots (Figure 4.17) along axis 1 and 2. Four environmental variables (litter depth, species richness of woody vegetation, cover of woody vegetation, and dominance of woody vegetation were all found to be significantly positively correlated with both axis 1 and 2 (Table 4.21). Litter depth and dominance of woody vegetation were recognised as the main factors correlated with the positioning of Araneida species on the ordination diagram. All four environmental variables mentioned above increased with increasing values along either axis. Evenness of woody vegetation was negatively correlated with axis 1; remnant study sites were found to have a less even distribution of woody species than planted restoration sites.

Table 4.21: Spearman rank correlation between variables and position of study plots on axis 1 and axis 2 for the ordination of Araneida data from all study plots (n = 26). The symbol * indicates a significant correlation at $\alpha = 0.05$. The critical value for df = 25 was $r = 0.337$.

Environmental Variable	Axis 1	Axis 2
Aspect	0.136	- 0.019
Slope	0.330	0.039
Litter depth	0.818*	0.554*
Species richness (vegetation)	0.380*	0.421*
Cover (vegetation)	0.507*	0.509*
H' (vegetation)	- 0.020	0.099
d (vegetation)	0.627*	0.610*
E (vegetation)	- 0.469*	- 0.091

4.4. Preliminary Discussion of Results

The preliminary discussion of results included here is limited in scope to the discussion of the results presented within this chapter. This discussion is shaped with respect to the objectives established at the outset of this ground active invertebrates chapter, with an additional section discussing the method of pitfall trapping. Placing the results within the context of the results obtained elsewhere in this study and other studies, in addition to determining how the results of this study fit within the framework of restoration success is undertaken in Chapter 6.

4.4.1. Did Species Composition Vary Between Planted Restoration and Remnant Study Sites?

4.4.1.1. All Invertebrates

A marked difference was apparent between invertebrate composition in planted restoration and remnant study sites. The ordination diagram illustrating the distribution of 'all invertebrate' species collected within the study sites (Figure 4.13) clearly demonstrates the large disparity between the composition of ground active invertebrate communities in these two types of study site. Study plots of remnant study sites were very tightly clustered together along axis 1 and axis 2, indicative of similar species composition within all remnant study sites. A large degree of separation present between study plots of planted restoration study sites occurred along both axes.

Diversity indices were used to reveal patterns of species abundance and heterogeneity between ground active invertebrates collected throughout the study area. Four of the

five diversity indices used for 'all invertebrates' indicated the presence of a significant difference between planted restoration and remnant study sites. Despite more invertebrate species being caught in pitfall traps within planted restoration study sites, no significant difference was detected in the mean number of species observed. Care is required when comparing absolute numbers of invertebrate between study sites (Crisp et al, 1998). Therefore, the total number of species observed was standardised to provide mean species richness values, per study site. This was done due to only two 10 × 10 m study plots being established in the P6 study site, in comparison with the three study plots established in each of the other eight study sites.

Within the 'all invertebrates' category, remnant study sites were found to contain significantly more invertebrate individuals than planted restoration study sites. Further, remnant study sites were also found to have significantly higher dominance values, reflecting a lesser dominance of a single, or few invertebrate species within these sites. The inverse pattern was found when investigating the heterogeneity and evenness of the distribution of trapped invertebrates. That is, planted restoration study sites were found to have a significantly more diverse array of invertebrates collected than remnant study sites. They were also found to have a significantly more even distribution of invertebrates than remnant study sites. These significant results further emphasise the apparent differences in abundance and composition of ground active invertebrates in the two types of study site.

4.4.1.2. Coleoptera

The ordination diagram for Coleoptera (Figure 4.16) was very similar to the 'all invertebrate' ordination. There was a clear separation of remnant study sites from planted restoration study sites. Again, remnant study sites were tightly clustered, suggesting a high degree of similarity between Coleoptera species found within this type of study site. The large separation between the two types of study site led to the suggestion that the P1 study site shared very few Coleoptera species in common with samples collected in any of the three remnant study sites. The relevant Jaccard's coefficients endorsed this assumption.

Fewer significant differences between planted restoration and remnant study sites were suggested for diversity indices, within the Coleoptera category. Planted restoration study sites were found to contain a significantly higher mean number of individuals than remnant study sites. The higher mean number of Coleoptera were the result of large numbers of a few Coleoptera species, as supported by the significantly lower dominance values resulting from the Berger-Parker index, than remnant study sites. As stated previously, low values of the Berger-Parker index here suggest the dominance of a lesser number of Coleoptera species in planted restoration study sites. No significant difference was apparent between the two types of study sites in terms of heterogeneity, or evenness of Coleoptera caught.

4.4.1.3. Araneida

The ordination of Araneida species by site (Figure 4.17) was less clear in the demarcation of study sites. A less obvious distinction between the two types of study site was evident. This observation corresponded with the results of the diversity indices for this category. Only one diversity index revealed a significant difference between the types of study site when comparing Araneida; the mean number of individuals. Planted restoration study sites were found to contain significantly more individuals than remnant study sites. Predacious species, such as some Araneida, are often some of the earliest colonisers of disturbed ground (Wheater & Cullen, 1997). The relative paucity of significant differences in this category may be due to the smaller sample size collected and difficulty in identification of distinct Araneida species.

4.4.2. Did Species Composition Vary Between Planted Restoration Study Sites?

4.4.2.1. All Invertebrates

Planted restoration study sites were separated by approximately 3 SD along both axis 1 and axis 2 in the ordination diagram showing the distribution of 'all invertebrates' (Figure 4.13). The positioning of the study plots of the P1 study site from those of the P4 and P5 study sites suggests that more than 50% species turnover took place from P1 to the latter two study sites. The Jaccard's similarity coefficients, using presence / absence data for 'all invertebrate' species substantiates this difference in invertebrate composition. The additional separation of the P5 and P6 study plots on the ordination

diagram by more than 1 SD, from which it can be inferred that these two study site also shared less then half their trapped invertebrate species in common, is also supported by the appropriate Jaccard's similarity coefficient. The ordination also suggested an apparent difference in invertebrate composition of the P1 and P2 study sites from the four other planted restoration study sites.

Diversity indices were again used to provide an indication of species of abundance and distribution among the six planted restoration study sites. A significant difference was detected in the mean number of invertebrate species located within planted restoration study sites. The results of the Tukey's test did not reveal the nature of this significant difference. This was thought to be due to the large amount of variation around the mean number of invertebrate species caught within each study site. No clear trend was observed for mean species richness to increase with age of study sites, due particularly to the low number of invertebrate species caught in the P6 study site pitfall traps.

The mean number of ground active invertebrate individuals was not found to differ between any of the six planted restoration study sites. Nor was evidence of significant difference suggested in values of the Berger-Parker index. A significant difference was apparent, however, in the heterogeneity of invertebrates trapped. P1 was found to have a significantly lower heterogeneity than P2 and P4 study sites. The P1 study site was also found to have the least even distribution of 'all invertebrates', while the P6 study site had the highest evenness value for planted restoration study sites. The difference in evenness values for these two study sites was found to be significant. The significant differences between P1 and the P4 and P6 study sites supports the observed separation between the study plots for each of these sites on the relevant ordination diagram (Figure 4.13). Study sites P1 and P5 were found to be the least similar, with a Jaccard's coefficient of $C_J = 0.21$.

4.4.2.2. Coleoptera

As stated in the previous section, the ordination diagram of Coleoptera species by site was very similar to the corresponding ordination of 'all invertebrates'. The same separation of P1 and P2 from the P3, P4, P5, and P6 study sites along axis 1 was

apparent. P2 was found to have a significantly higher mean number of Coleoptera species than P1, P5 and P6. In addition, P1 had a significantly lower mean number of Coleoptera species trapped than P3 and P4. P1 was also found to have significantly lower values for the Shannon index than P4 and P2. No significant difference was apparent in the mean number of Coleoptera individuals caught in planted restoration study sites. Nor was there a suggestion of a significant difference in the dominance of evenness of distribution of trapped Coleoptera species.

4.4.2.3. Araneida

The ordination of Araneida species by site (Figure 4.17) did not indicate a clear separation of planted restoration study plots along either axis 1 or axis 2. Similarly, statistical analysis of the diversity indices used did not reveal a significant difference between any of the six study plots. The P4 and P5 study sites were found to share the most species of Araneida in common with a Jaccard's coefficient of $C_J = 0.80$.

Simmonds et al (1994) measured the composition of spider communities, in order to assess the success of restoration of mine sites in Western Australia. This study reported a trend in the development of Araneida community composition associated with habitat development (a function of site age). Simmonds et al (1994) found that the number of spider species present in the rehabilitated sites was only loosely tied to the age of the rehabilitation. At Cape Foulwind, the three-year old site did have the lowest species richness, indicating that spider species richness was initially low.

4.4.3. Was the Distribution of Ground Active Invertebrates Between Study Sites Related to Environmental Variables?

A number of complex biotic and abiotic interactions have been influential in determining the distribution of ground active invertebrates across the study area. A study by Simmonds et al (1994), investigating the relationship between various habitat features and Araneida communities present found that litter depth, vegetation cover and density all had a significant positive influence on recolonisation. Soil moisture, light intensity, and ground litter were identified as major environmental attributes affecting the distribution of carabids (ground beetles) in northern Germany (Antvogel & Bonn, 2001). Magura et al (2001) found relative humidity, ground temperature and vascular plant cover to be the most important factors affecting

Coleoptera distribution along forest-grassland transects in Hungary. Latitude, canopy structure, soil organic matter and vegetation density were found by Jukes et al (2001) to be the most significant factors affecting the diversity of Coleoptera. The findings of this study concur with the above, as litter depth and vegetation cover were found to be significantly correlated with the positioning of invertebrates on the ordination diagrams, for all three categories. Of the environmental variables investigated, litter depth was determined to be the primary factor responsible for the separation of planted restoration and remnant study sites for all three categories ('all invertebrates', Coleoptera, and Araneida).

The development of invertebrate communities may be inextricably linked to the degree to which vegetation is established (Magura et al, 2001; Wheater & Cullen, 1997). Differences in invertebrate distribution found between planted restoration and remnant study sites may reflect the degree to which vegetation is established. For example, Morris (2000) suggests that invertebrates of grassland are characterised by adaptations to a relatively simple structural system, tending to demonstrate features such as opportunism. Araneida communities were found to change as vegetation matured, from a dominance of pioneer species to a community of Araneida requiring less harsh conditions in a study by Simmonds et al (1994). Reay & Norton (1999a) found that the provision of adequate floristic and structural diversity was important to allow for the full range of native invertebrate species to colonise during restoration. Magura et al (2001) emphasise that carabids depend more on habitat structure than on the presence of specific plant species. Vegetation composition, therefore, may not have as strong an influence on the activity of invertebrates as many ecologists presume (Hutcheson & Kimberley, 1999).

Habitat structure may affect the likelihood of an invertebrate being caught, resulting in biased data for studies aiming to determine differences in community structure between locations with different habitat structure, as in this study, or when habitat structure changes over time (Melbourne, 1999). Habitat structure is the spatial configuration comprising an invertebrate's environment, particularly the environment immediately surrounding the pitfall trap (Melbourne, 1999) It must be remembered that habitat structure is applicable to both above and below ground structure (Morris,

2000). The effect of habitat structure on the likelihood of a ground active invertebrate being caught in a pitfall trap was first investigated by Greenslade (1964). Habitat structure can affect the likelihood of trapping via microclimatic effects, habitat dilution, and the response of species to the physical features of the environment, such as the density of the vegetation, or the roughness of the soil surface, whether these factors act independently or concurrently (Magura et al, 2001; Melbourne, 1999).

Habitats with a more complex structure have more surface area available for invertebrates to move around on, thereby reducing the effective number of pitfall traps per unit area; the habitat dilution hypothesis (Melbourne, 1999). For example, in dense vegetation, ground active invertebrates may not only move around on the ground, but also on the vegetation. The importance of this dilution effect will be related to the scale at which the invertebrate perceives its environment. The dense litter layer and increased frequency of broken branches, on the ground in remnant study sites may have reduced the effective number of pitfall traps. Crist & Ahern (1999) found differences in vegetation structure affected beetle movement, thereby affecting the capture rate in pitfall traps. Melbourne (1999) found a tendency for capture probabilities to increase as habitats became more open, however, this is assuming that the frequency of occurrence data accurately reflect the probability of capturing a ground active invertebrate when present.

The relative abundance and quality of food sources for Coleoptera are an additional consideration influencing their distribution (McElrea, 2002). Predatory Coleoptera species dominated planted restoration study sites, both in terms of species and the number of individuals caught. This dominance of predatory Coleoptera is thought to be due to the paucity of ground litter within these study sites, resulting in a lack of appropriate niche sites. Dense vegetation should provide a greater food source for herbivorous species, and higher levels of detritus (Wheater & Cullen, 1997), possibly accounting for the greater proportion of herbivorous Coleoptera detected in remnant study sites.

It appears that all planted restoration study sites still have quite some structural development necessary before they can be directly comparable to remnant study sites

(Chapter 3), due to the apparent effect of site age on the likelihood of being caught in pitfall traps. It is recognised that looking for trends with age (time since planting) within the planted restoration study sites is not strictly appropriate, as that would assume that age is the only source of variation between the study sites. The study sites used in this research project were not all based on the same substrate. This may have resulted in differing invertebrate communities forming, confounding any age effect. Halsall & Wratten (1988) found however, that capture rates were not affected by differing substrates, which may imply that invertebrates caught in the pitfall traps, at least, were not affected by this additional variation.

4.4.4. Use of Pitfall Traps

Pitfall traps provide a convenient and effective means of obtaining a continuous record of ground active invertebrates and provide a useful indication of the presence and absence of species in various habitats (Melbourne, 1999; Crisp et al, 1998; Simmonds et al, 1994; Topping & Sunderland, 1992; Halsall & Wratten, 1988). Pitfall trapping is a continuous sampling method, not vulnerable to problems associated with spot sampling in time (Topping & Sunderland, 1992). They are a commonly used method to determine whether differences exist in population size or community structure, either spatially or temporally (Melbourne, 1999). Pitfall traps have been used extensively to provide information on the number and activity of ground active invertebrates, particularly Coleoptera (Magura et al, 2001) and Araneida (Simmonds et al, 1994; Halsall & Wratten, 1988).

Despite their prevalence in the literature, pitfall trapping is a method that has been heavily criticised (Topping & Sunderland, 1992). Southwood (1978) stated that careful attention must be paid to the potential sources of error, while Adis (1979) went further, listing a total of eighteen potential sources of error (Topping & Sunderland, 1992). Pitfall traps are selective in their catch as numbers caught are density dependent (Crisp et al, 1998) and the level of activity of various species within that population (Melbourne, 1999; Luff, 1996; Greenslade, 1964). Activity levels can be affected, independently of population density, by a variety of factors, such as weather and habitat structure, leading to biased estimates of population density (Melbourne, 1999). Adis (1979) noted that climatic conditions of the experimental area, such as

soil humidity, precipitation, and temperature could affect activity levels of invertebrates resulting in variation in catch size. Invertebrate size, in addition to activity, has been shown to be an important factor influencing catch rate (Halsall & Wratten, 1988). Greenslade (1964) attributed this to the fact that larger, faster moving invertebrates cover greater distances, the consequence of which is an increase in pitfall trap encounters. Standen (2000) reports that pitfall traps capture greater numbers of larger carabids and under-represent smaller species. However, Halsall & Wratten (1988) found that interspecific differences in capture rate were related to the ability of invertebrates to perceive the trap edge, rather than activity levels.

Not all invertebrate species that are active on the ground are caught by pitfall traps (Adis, 1979). Pitfall trapping results in the under-sampling of invertebrates that live on vegetation as well as flying species (Buckton & Ormerod, 1997). A more comprehensive study of invertebrate diversity would require sampling of invertebrates from other strata, such as the canopy and shrub layers within the study area (Oliver & Beattie, 1996).

Ground invertebrate communities undergo a continual variation of populations and species turnover over the duration of the adult Coleoptera activity period from spring to autumn (Hutcheson & Jones, 1999; Hutcheson, 1990; Moeed & Meads, 1987). Pitfall trapping would ideally occur over the duration, or if not possible, several times for short periods (not less than four weeks) over this period (Adis, 1979). Sampling for the entirety of this period would have provided useful insight into the relative temporal diversity of ground active invertebrates within the study area. However, practical constraints restricted the length of time available for the collection of invertebrate data. Consequentially, a six week period from early December 2001 through to mid January 2002 was chosen. Hutcheson & Kimberley (1999) have endorsed early summer catches as being the most characteristic of sites in New Zealand. The time period chosen to undertake pitfall trapping appears to have coincided with a period of high invertebrate activity as the catch was diverse and contained numerous individuals (over 7 000 total). Because catches of pitfall traps are dependent on the activity levels of ground active invertebrates, numbers caught can be

interpreted as being representative of the catch only, not of the relative numbers of invertebrates present within the study area as a whole (Halsall & Wratten, 1988).
It is assumed that biases in capture rates were the same between the study plots within each study site. This assumption is reasonable as pitfall trapping took place continuously for an equal duration, in areas of similar vegetation structure, and weather patterns would be the same for all study plots in the same study site (Melbourne, 1999). This critical assumption cannot be made with confidence between study sites, however, as litter depth and groundcover varied greatly. In planted restoration study sites ground cover was primarily exotic grasses, while a dense litter layer was present in remnant study sites. Microclimatic conditions, exacerbated by the type of ground cover, such as soil temperature, relative humidity, soil moisture, although not recorded directly will have been a further source of variation affecting the effective number of pitfall traps and capture rates of ground active invertebrates, accordingly.

In addition to an increased sampling period for invertebrates (i.e. over the entire summer to autumn period), measurement of microclimatic variables such as relative humidity, ground temperature, soil moisture and vegetation structure would provide a more comprehensive insight into communities of ground active invertebrates in the study area.

4.4.5. What is the Potential of Ground Active Invertebrates to be used as Environmental Indicators for Restoration Success?

In order for ground active invertebrates to be regarded as appropriate environmental indicators, taxa should meet various criteria. Indicator taxa should be functionally important and closely interrelated with other aspects of the ecosystem, abundant, diverse and widespread. It is also important that environmental indicators can be sampled and identified readily, and provide interpretable results (Andersen, 1990). The relationships that indicator taxa have to the diversity of other taxa should be known, and they should respond in a predictable way to environmental parameters and disturbance events, providing insights into the nature of, and underlying mechanisms of such change (Oliver & Beattie, 1996; Andersen, 1990). The use of invertebrates as environmental indicators is most effective when supported by a

predictive understanding of community composition in relation to environmental stress and disturbance (King et al, 1998).

Ants are widely used as environmental indicators, particularly in relation to minesite restoration (King et al, 1998; Andersen & Sparling, 1997). Due to their high number and diversity, ecological importance at all trophic levels, well understood community dynamics, and ease of sampling, ants are considered an attractive target taxa to act as environmental indicators (Andersen & Sparling, 1997). Following disturbance, changes in ant community structure have been shown to reflect changes in many other invertebrate groups (King et al, 1998; Andersen & Sparling, 1997). Carabids are also useful environmental indicators due to their sensitivity to environmental conditions and rapid responses to habitat change (Abildsnes & Tømmerås, 2000).

Several approaches can be taken to using ground active invertebrates as environmental indicators. One approach is to consider invertebrates at an 'ordinal' level, such as Coleoptera as a whole. The disadvantage of this is it fails to separate species with different ecological requirements or functions within the ecosystem. The presence of large numbers of Coleoptera after a restoration effort, as was found in this study, will not necessarily mean that an appropriate complement of Coleoptera have been restored (Majer et al, 2002).

The use of one or more taxa as surrogates for a range of other invertebrates is another approach. It has been observed that variations in richness of certain invertebrate groups are not necessarily correlated across sites and may not be correlated with floristic diversity. Majer et al (2002) note that trends across sites tend to be more concordant when invertebrate communities of highly disturbed sites, such as restored minesites, are compared with each other and with undisturbed benchmark areas. The most satisfactory approach, when using invertebrates as environmental indicators is to use a range of invertebrates, representing organisms associated with a complementary range of ecological functions such as nutrient cycling, soil structuring, and herbivory (Majer et al, 2002). The larger the suite of species used as indicators, the more powerful the ability to discriminate between habitats and communities will be (Harris & Burns, 2000).

To be beneficial to Holcim's restoration effort, a selection of ground active invertebrates with varying functional roles should be used as environmental indicators. Future comparisons of invertebrate communities, looking for developmental trends of sites tending towards the composition and abundance of ground active invertebrates found in remnant study sites would be beneficial. By including ground active invertebrates in this study, a benchmark has been provided, to allow future comparisons to be made in a statistically robust manner.

4.5. Summary

A large degree of dissimilarity was apparent in ground active invertebrate communities caught in planted restoration sites and the benchmark remnant study sites. This difference was particularly evident for 'all invertebrates' and Coleoptera, but less distinct for Araneida. Significantly greater numbers of Coleoptera were collected within remnant study sites than planted restoration study sites. Coleoptera that were collected in both types of study site revealed a difference in functional groups. Greater numbers of herbivorous Coleoptera were located within remnant study sites, while predatory Coleoptera dominated in planted restoration study sites. The contrary was found for Araneida, where significantly more Araneida were collected within planted restoration than remnant study sites. The ordination diagrams suggested an apparent difference in invertebrate composition of the P1 and P2 study sites from the four other planted restoration study sites. This difference was substantiated by diversity indices. Litter depth, of the environmental variables investigated, was found to be the key driver of invertebrate distribution over the nine study sites. A range of invertebrates, associated with complementary ecosystem functions, should be used as environmental indicators by Holcim to aid future assessments of restoration success at Cape Foulwind.

5. ECOSYSTEM ATTRIBUTES

5.1. Introduction

Successful restoration cannot be achieved without the formation of a self-sustaining, fully functioning ecosystem. The purpose of this chapter is to investigate a variety of ecosystem attributes essential to the establishment of such an ecosystem. Through the study of these ecosystem attributes, baseline values will be determined, upon which future comparisons can be based.

This chapter investigates soil attributes, seed rain, ground litter (including litter depth / volume and litter decomposition) and comparison of light in terms of visible sky in each of the study sites. The particular objective, established at the outset of this study, pertinent to this chapter was to determine whether the ecosystem processes of litter decomposition and seed dispersal have occurred within the planted restoration study sites. All assessments done in this chapter aim to determine whether:

1. a significant difference occurs between values obtained in remnant and planted restoration study sites for ecosystem attributes;
2. ecosystem attributes and processes discussed differ among the six planted restoration study sites.

5.2. Methods

5.2.1. Field Methods

All data relating to ecosystem attributes were collected after pitfall traps (see Chapter 4) had been removed from the field, so as not to interfere with the results of the ground active invertebrate aspect of the study.

5.2.1.1. Ground Litter

Litter Depth / Volume

Litter depth measurements were undertaken within 10 × 10 m study plots (Section 4.2.1.) throughout the study area. As mentioned previously, all study sites contained three such study plots, except the P6 study site, in which, due to its restricted size,

only two study plots were established. Within each study plot, nine litter depth measurements were taken, to the nearest millimetre. The nine measurements were taken in a cross design within each study plot as illustrated in Figure 5.1.

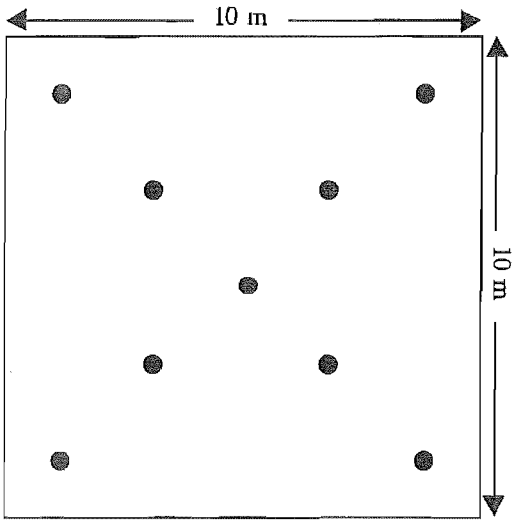


Figure 5.1: Location of ground litter measurements within each of the twenty six 10 × 10 m study plots.

Nine samples of litter were taken from each 10 × 10 m study plot. Each sample was collected from a 20 × 20 cm quadrat, at the position used for obtaining litter depth measurements (Figure 5.1). Excess materials, such as sticks and stones were removed from each sample. Samples were subsequently placed in labelled paper bags and oven dried for 48 hours at 70°C. Following their return to ambient laboratory conditions, samples were removed from the paper bags and individually weighed.

Litter Decomposition

Litter bags (10 × 15 cm), made from polythene mesh were used to investigate litter decomposition in the nine study sites (Moro & Domingo, 2000; Cortez, 1998). Two mesh sizes were used, 3 mm on the exposed surface of the litterbag, and 1 mm mesh on the underside of the litterbag. The two different sized mesh materials were used to retain as much of the contents of the litterbag as possible, while impeding the movement of ground active invertebrates involved in the process of litter decomposition as little as possible. In mid-April 2002, litter samples were collected from numerous *Coprosma robusta* plants throughout the study area. The affect of microclimate on litter decomposition was of interest rather than the absolute decomposition rates themselves, therefore no attempt was made to use a mixture of

species, or to collect naturally abscised leaves (Didham, 1998). The litter collected was thoroughly mixed and air-dried in the laboratory. Samples were then weighted and approximately 2 g (1.96 – 2.04 g) of litter was placed in each litterbag. Litter samples were not oven dried prior to being placed in the field as that may have induced chemical changes, influencing decomposition (Belyea, 1996).

Six replicates were placed into each of the twenty-six 10 × 10 m study plots (Figure 5.1) in randomly selected positions in mid-April, 2002. Care was taken to place the litter bags as close as possible to the soil surface without causing excessive disturbance to the existent litter layer (Hector et al, 2000). The litter bags were held in place using nails, with wire mesh (13 mm mesh) was placed over the top, secured using six hooks made from 1.6 mm wire. Two litter bags were removed from each study plot after three, six, and nine months in the field.

A further twenty samples of *C. robusta* leaves were weighed and dried at 70°C in order to obtain a wet weight : dry weight ratio, to use as the mean initial weight for data analysis.

Upon collection, residual litter was removed from the litter bags, and the wet weight recorded after removing any soil material and grass shoots (Hector et al, 2000). The litter samples were then placed in labelled paper bags and oven dried at 70°C for 48 hours. The samples were then removed from the bags and individually weighed.

5.2.1.2. Soil

Three soil samples were taken at a depth of 15 cm from randomly selected positions from each of the twenty six study plots (section 4.2.1.). Each soil sample was subsequently oven dried, in a paper bag, for seven days at 70°C. Once samples returned to ambient laboratory conditions, the soil samples were sieved to <0.5 mm size, then weighed, with 20 g of each sample retained. The three samples from each study plot were then combined into one and then sent for analysis at Landcare Research in Palmerston North, New Zealand. The number of samples that could be analysed was limited due to financial constraints of this study. Each soil sample was analysed for pH (water), total carbon, total nitrogen, Olsen phosphorous, cation

exchange capacity, exchangeable bases (calcium, magnesium, potassium, and sodium), and percent base saturation. The methods used to determine each soil attribute have been taken from the Landcare Research website (2003).

pH

8 g of soil was mixed to a slurry with 20 mL of deionised water (or 0.01M $CaCl_2$ or M KCl) and left to stand overnight. The pH was then measured using a combination electrode (Landcare Research, 2003).

Total Carbon and Nitrogen

To determine levels of total carbon and total nitrogen, the soil samples were heated in a stream high purity oxygen in a Leco furnace to produce CO_2, N_2 and NO_x. A subsample of the combustion gases was then passed through a heated copper catalyst that further reduced the NO_x to N_2, which is then measured by thermal conductivity. The CO_2 was measured with an infrared detector (Landcare Research, 2003).

Olsen Phosphorous

The method involved in determining the level of available phosphorous is based on the phosphorous extraction method of Olsen et al (1984), as described by Blakemore et al (1987). The method used an extraction with bicarbonate to estimate plant available phosphorous in the soil (0.5M sodium bicarbonate, pH 8.5, 1:20 soil:extractant, 30 minutes shaking). The level of phosphate in the extracts was determined by utilising a colorimetric method on a Lachat flow injection analyser (Landcare Research, 2003).

Cation Exchange Capacity

Cation exchange can be described as the interchange between a cation in solution and another cation on the surface of a surface-active material, such as clay or organic matter (Landcare Research, 2003). The principle cations found in an exchangeable form are the bases calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+) and sodium (Na^+), and the cations hydrogen (H^+) and aluminium (Al^{3+}). Landcare Research (2003) define cation exchange capacity (CEC) as the total amount of cations that the soil can retain. It is measured by leaching the soil with an excess of a particular cation.

Landcare Research (2003) used the automatic extractor procedure described by Blakemore et al (1987). In this method, exchangeable bases are removed and exchange sites are saturated with ammonium ions by leaching with neutral molar ammonium acetate (Landcare Research, 2003). Excess ammonium acetate was then washed from the sample with alcohol. The adsorbed ammonium ions are then displaced from the exchange sites by leaching the ample with molar sodium chloride (1M NaCl) (Landcare Research, 2003). The amount of ammonium in the sodium chloride solution, to give the CEC was done using a colorimetric method on a Lachat flow injection analyser (Landcare Research, 2003).

Exchangeable Bases

Determination of the exchangeable bases (calcium, magnesium, potassium and sodium) is useful as these are the forms of the elements that are considered to be available to plants, by exchange with hydrogen ions from the exudates of root hairs and soil microorganisms, hence they are important for plant growth (Landcare Research, 2003). Because CEC and percent base saturation were also requested from Landcare Research, a leaching by automatic extractor procedure was used. The leaching by automatic extractor procedure is described in the above explanation for CEC.

Percent Base Saturation

Because information on CEC and exchangeable bases were requested from Landcare Research (2003), a leaching by automatic extractor procedure was used. The leaching by automatic extractor procedure is described in the explanation for CEC. The proportion of the CEC that is occupied by exchangeable bases (referred to as percent base saturation) is regarded as a useful fertility index often used in soil classification systems (Landcare Research, 2003). Percent base saturation is determined using:

$$\%BS = \frac{TBS\left(cmol_c kg^{-1}\right)}{CEC\left(cmol_c kg^{-1}\right)} \times 100$$

where, $\%BS$ is the percent base saturation, TBS is the total exchangeable bases in the soil, and CEC is cation exchange capacity.

5.2.1.3. Seed Rain

Seed rain was measured using twenty six seed traps. A single seed trap was established at the centre of each 10 × 10 m study plot (Section 4.2.1.). Each trap consisted of a wire ring, 50 cm in diameter, with dense shade cloth attached around the circumference of the ring. The wire ring was supported by three wooden stakes, so that it sat 1.5 m above ground level in the remnant study site plots, and 0.6 – 0.8 m above ground level in study plots located in the planted restoration sites. Seed traps located in the latter study sites were encircled by wire mesh (13 mm mesh size) to prevent damage to experimental set up by birds, primarily *Gallirallus australis* (weka). The seed traps were located at different heights in the two types of study site because of the lower canopy level in the planted restoration study sites. Because seed traps were 1.5 m above the ground in remnant study sites, the risk of damage by wekas was greatly diminished; consequently the protective wire mesh was not necessary. The seed traps were established so that the wire ring was horizontal. The shade cloth was tied firmly at its base, and hung so that it formed a collecting cup that was 60 cm deep, with a catch area of 0.196 m^2. The maximum mesh size of the shade cloth formed a triangle of 1 mm^2, but due to the arrangement of the cloth beneath the wire ring, and the bunching of the cloth in the base of the seed trap, the effective mesh size was considerably smaller (Dungan et al, 2001).

Seed rain was collected for approximately five months during 2002. Seed traps were installed in the study plots during the second week of January and left until the end of May. During this period, the seed trap were cleared of any obstructing materials, such as branches or fern fronds, but were only emptied of their seeds once, at the completion of this period. The trapping period was designed to coincide with the main period of seedfall for most species (Burrows, 1994).

Upon collection, seeds were separated from other materials and stored in labelled paper bags until identification could be undertaken. Seeds were identified with the aid of Webb & Simpson (2001), with their frequency recorded for analysis.

5.2.1.4. Light (Visible Sky)

Digital photographs were taken at ground level from five points in each of the twenty six study plots (Figure 5.2). The camera was held in place using a self-levelling mount (type SLM2), designed to aid a camera and fisheye lens to remain aligned to the horizon and North (Rich et al, 2000). The Nikon Coolpix 900 series camera with a hemispherical lens attachment was used, as was necessary to enable use with HemiView canopy analysis software (see Section 5.2.2.4.). Delta-T Services developed the HemiView software; it is a Windows-based program designed for convenient image analysis of hemispherical photographs (Rich et al, 2000).

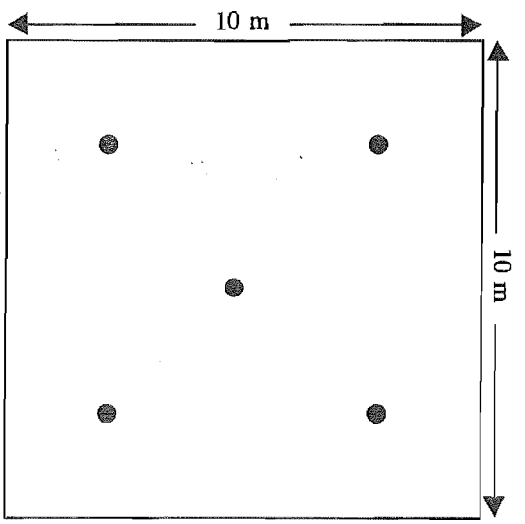


Figure 5.2: The five positions from which photographs were taken within each of the twenty six 10 x 10 m study plots.

5.2.2. Data Analysis

5.2.2.1. Ground Litter

Litter Depth / Volume

Litter depth and volume values were compared statistically between study site types and between the six planted restoration study sites using ANOVA. The statistical package SAS V8 was used. Pairwise multiple comparisons were also undertaken using Tukey's test to determine the nature of the significant differences detected. The level of significance was set at $\alpha = 0.05$.

Litter Decomposition

Litter decomposition was assessed in terms of percent mass loss based upon mean dry weights established at the beginning and end of the experimental periods (Hector et al, 2000). Mean levels of percent mass loss at the end of the three, six and nine month periods were compared statistically with general linear models (GLM), using the statistical package SAS V8. Pairwise multiple comparisons were run using Tukey's test to determine the nature of significant differences detected by the GLMs. The level of significance testing was set at $\alpha = 0.05$.

5.2.2.2. Soil

Upon receiving output from Landcare Research in Palmerston North, New Zealand where soil analysis was undertaken, results were tested for significant differences in the mean levels of pH (water), total carbon (%), total nitrogen (%), Olsen phosphorous (mg/kg), cation exchange capacity (cmol(+)/kg), calcium (Ca^{2+}) (cmol(+)/kg), magnesium (Mg^{2+}) (cmol(+)/kg), potassium (K^{+}) (cmol(+)/kg), and sodium (Na^{+}) (cmol(+)/kg), and base saturation (%), between study sites.

Mean levels of the above were compared statistically using one-way analysis of variance (ANOVA), using the statistical package SAS V8. Pairwise multiple comparisons were run using Tukey's test to determine the nature of significant differences detected by ANOVA. The level of significance testing was set at $\alpha = 0.05$.

5.2.2.3. Seed Rain

Mean values of species richness of seed rain were compared statistically using ANOVA, with the aid of the statistical package SAS V8. Pairwise multiple comparisons were also done using Tukey's test to determine the nature of significant differences detected. The level of significance was set at $\alpha = 0.05$.

5.2.2.4. Light (Visible Sky)

The mean amount of sky visible within each study site was compared statistically between type of study sites (remnant or planted restoration) and between the six planted restoration study sites using ANOVA. The statistical package SAS V8 was used. Pairwise multiple comparisons were also done using Tukey's test to determine

the nature of the significant differences detected. The level of significance was set at $\alpha = 0.05$.

5.3. Results

5.3.1. Ground Litter

5.3.1.1. Litter Depth

Litter depth was found to vary significantly with the type of study area (F = 193.71, df = 1, P = < 0.001). The Tukey's test revealed that remnant study sites had a significantly deeper litter layer than planted restoration study sites (Table 5.1). An ANOVA investigating mean litter depth between each of the six planted restoration study sites suggested that a significant difference in litter depth was present (F = 10.64, df = 5, P = < 0.001). According to the Tukey's test, the mean depth of litter layer in P6 was significantly greater than that in the P1, P2, P4, and P5 study sites. Additionally, the P3 study site was found to have a litter layer significantly deeper than that of P1, P4 and P5.

No evidence of ground litter was apparent in the P1 study site. P6 had the litter layer of the greatest depth of the planted restoration study sites. Average litter layers in remnant study sites were recorded as at least on and a half times the depth of litter layers in planted restoration study sites. The study site with the greatest mean depth of litter was R3, with 0.18 m (Figure 5.3).

Table 5.1: Descriptive statistics of litter depth. Part (a) investigates the difference in litter depth between remnant sites and planted restoration sites, while part (b) compares litter depth values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	81	0.06	0.59	0.17^{A}	0.01
	P	153	0.00	0.32	0.04^{B}	0.00
(b)	P1	27	0.00	0.00	0.00^{C}	0.00
	P2	27	0.00	0.22	0.04^{B}	0.01
	P3	27	0.00	0.22	0.08^{AB}	0.01
	P4	27	0.00	0.19	0.02^{C}	0.01
	P5	27	0.00	0.15	0.02^{C}	0.01
	P6	18	0.03	0.32	0.10^{A}	0.02

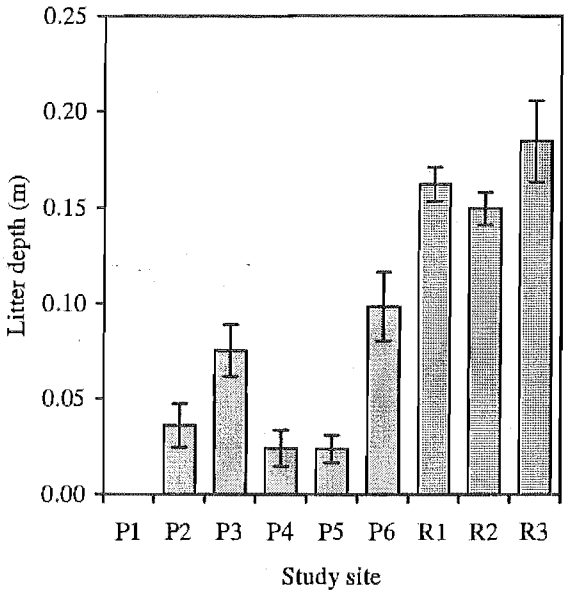


Figure 5.3: Mean litter depth for each of the nine study sites. The error bars depict the standard error around the mean.

A general tendency for regeneration density to increase with litter depth was apparent. Figure 5.4 illustrates this point, with a linear pattern evident. Regeneration density was defined as the mean number of regenerating seedlings identified within each study site.

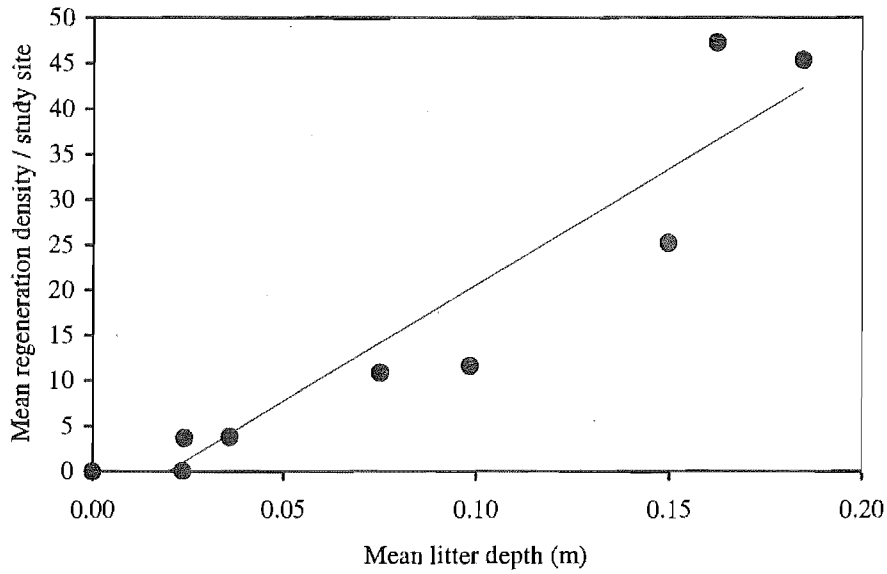


Figure 5.4: Comparison of regeneration density and litter depth.

An ANOVA run for litter volume by type of study site (planted restoration or remnant study site) suggested the presence of a significant difference (F = 242.28, df = 1, P = < 0.001). According to the Tukey's test, planted restoration study sites had significantly less litter volume than remnant study sites (Table 5.2). A significant difference in litter volume was also detected between the six planted restoration study sites (F = 8.81, df = 5, P = <0.001). The Tukey's 'comparison of means' test revealed that P6 had significantly greater average litter volume than the five other planted restoration study sites. R3 had the greatest amount of litter, on average, for all the study sites (14.93 g), while no evidence of litter was found within the P1 study site (Figure 5.5). P6 had the largest volume of litter for the planted restoration study sites (5.00 g).

Table 5.2: Descriptive statistics of litter volume. Part (a) investigates the difference in litter volume between remnant sites and planted restoration sites, while part (b) compares litter volume values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	81	0.00	33.52	11.83^{A}	0.79
	P	153	0.00	19.04	1.31^{B}	0.26
(b)	P1	27	0.00	0.00	0.00^{B}	0.00
	P2	27	0.00	0.68	0.18^{B}	0.05
	P3	27	0.00	15.55	2.23^{B}	0.88
	P4	27	0.00	4.48	0.48^{B}	0.22
	P5	27	0.00	5.79	1.21^{B}	0.34
	P6	18	0.00	19.04	5.00^{A}	1.36

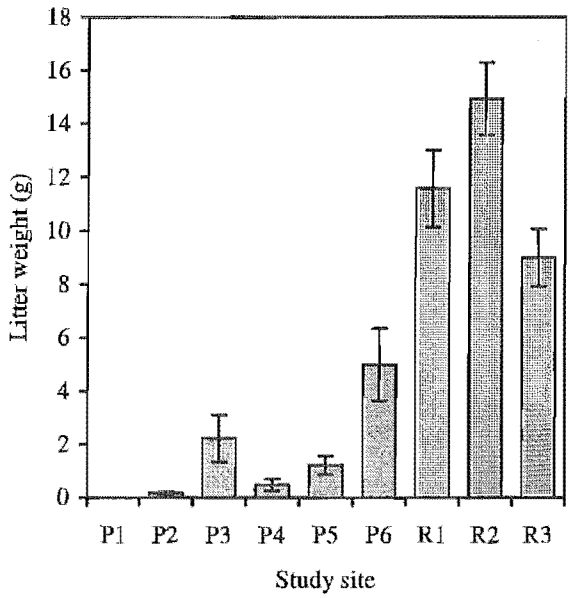


Figure 5.5: Mean litter volume for each of the nine study sites. The error bars depict the standard error around the mean.

A general tendency for regeneration density to increase with litter volume was apparent. Figure 5.6 illustrates this point, with a clear linear pattern evident.

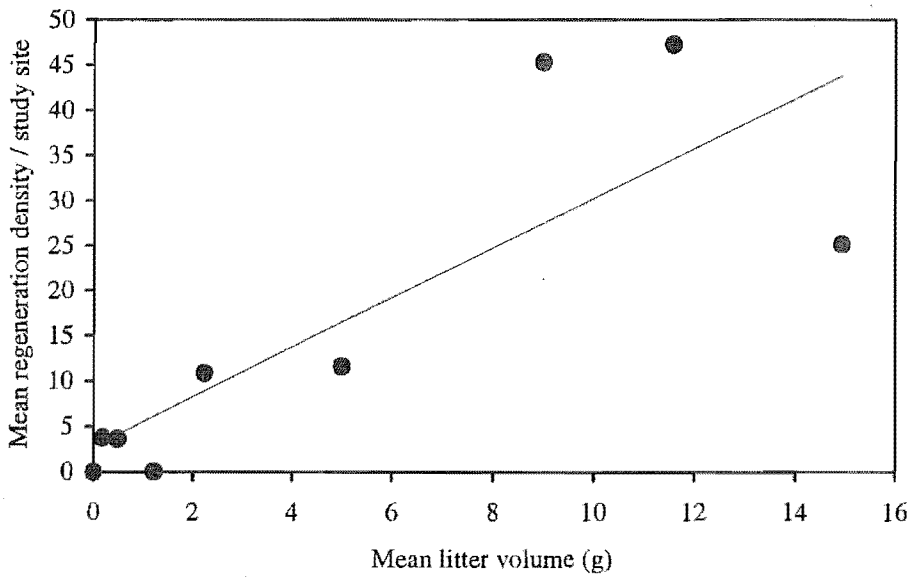


Figure 5.6: Comparison of regeneration density and litter volume.

5.3.1.2. Litter Decomposition

Litter decomposition was measured as the percent mass loss from a mean original dry weight. The greatest mass lost in the three month period occurred in P2 (Figure 5.7), with a mean loss of 85.1%. Litter decomposition appeared to be the slowest, of all study sites, in the P6 site, with a loss of 54.3% from the original mean dry weight. An ANOVA run to investigate the difference within the three month period showed no significant difference between the percent weight loss of litter placed in remnant study sites and in planted restoration study sites ($F = 0.80$, df =1, $P = 0.376$). Neither was a significant difference found between mean percent mass lost in the six planted restoration study sites ($F = 2.09$, df = 5, $P = 0.096$) (Table 5.3).

Table 5.3: Percent weight loss from mean initial dry weight after three months in the field. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	16	30.9	87.1	59.1^{A}	4.4
	P	32	16.9	98.7	71.9^{A}	3.8
(b)	P1	6	27.0	91.8	68.6^{A}	11.5
	P2	6	57.8	98.7	85.1^{A}	7.5
	P3	6	45.1	98.5	73.1^{A}	8.8
	P4	6	30.0	90.0	61.8^{A}	9.1
	P5	6	76.1	91.5	84.3^{A}	2.3
	P6	6	16.9	71.8	54.3^{A}	12.6

The largest amount of mass lost due to litter decomposition over a six month period was again found to occur in litter bags located in the P2 study site, with a mean loss of 91.4% of the original mean dry weight. The least amount of mass loss occurred in P1, which lost a mean 57.5% of its original mass (Figure 5.7). No significant difference was apparent in the weight loss that occurred in remnant and planted restoration study sites ($F = 0.11$, df = 1, $P = 0.740$). Similarly, an ANOVA run to determine whether a significant difference existed between the six planted restoration study sites suggested that no such significant difference was present ($F = 0.11$, df = 5, $P = 0.990$) (Table 5.4).

Table 5.4: Percent weight loss from mean initial dry weight after six months in the field. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	16	72.0	97.5	89.7^{A}	2.4
	P	32	22.4	97.5	81.9^{A}	3.7
(b)	P1	6	22.4	85.3	57.5^{A}	18.5
	P2	6	86.0	97.5	92.6^{A}	2.3
	P3	6	53.3	91.3	77.9^{A}	8.7
	P4	6	74.3	94.2	84.8^{A}	5.0
	P5	6	70.5	95.5	84.5^{A}	5.8
	P6	6	81.8	95.2	88.5^{A}	6.7

An ANOVA run to determine whether a significant difference occurred in the mean amount of mass lost over a nine month period in the field between the two types of study site (planted restoration and remnant) suggested that no such significant difference occurred ($F = 2.43$, df = 1, $P = 0.125$). Like all other comparisons of litter decomposition, no significant difference was detected between mean amounts lost between planted restoration study sites over the nine month period ($F = 1.04$, df = 5, $P = 0.416$). The largest decrease in mass over a nine month period in the field occurred in the P4 study site, which had a 91.2% loss. The smallest amount of mass lost in planted restoration study sites occurred in P5, losing 84.9% of the original mass (Figure 5.7). However, the least amount of weight loss overall occurred in R3 with a mean loss of 84.0% (Table 5.5).

Table 5.5: Percent weight loss from mean initial dry weight after nine months in the field. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	16	62.3	96.8	85.9^{A}	2.1
	P	32	58.1	97.3	88.0^{A}	2.1
(b)	P1	6	82.5	93.5	88.7^{A}	2.5
	P2	6	83.5	96.8	90.6^{A}	3.9
	P3	6	62.4	96.5	86.2^{A}	8.0
	P4	6	83.6	97.3	91.2^{A}	4.0
	P5	6	58.1	96.7	84.9^{A}	6.9
	P6	6	84.5	95.9	88.9^{A}	2.7

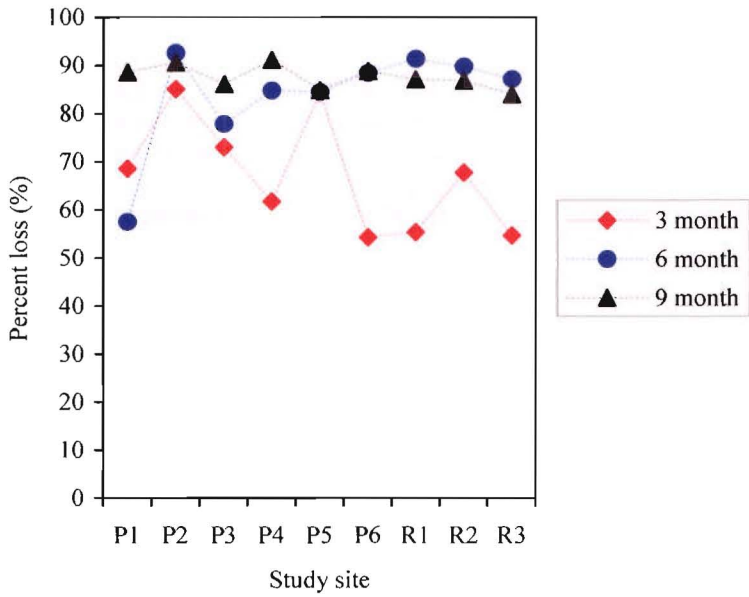


Figure 5.7: Percent mass of litter lost through decomposition from estimated original mass values.

5.3.2. Soil

pH

No significant difference was detected in pH levels of soil samples from remnant and planted restoration study sites ($F = 0.09$, df = 1, $P = 0.772$). All soils were found to be acidic. The most acidic soil samples were taken from the P2 study site, the least acidic from the P1 study site (Figure 5.8). A significant difference in pH levels was suggested between soil samples of the six planted restoration study sites ($F = 23.99$, df = 5, $P = <0.001$). Soil samples from the P2 and P4 study sites were shown, by the Tukey's test, to be significantly more acidic than those from the P1, P5 and P6 study sites. The P3 soil samples were found to have significantly more acidic soil samples than P1 and P5 (Table 5.6).

study site. Soil samples from P2 had the greatest total carbon values within planted restoration study sites (Figure 5.9).

Table 5.7: Descriptive statistics of total carbon values of soil samples collected. Part (a) investigates the difference in carbon levels between remnant and planted restoration study sites, while part (b) compares carbon values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	4.0	9.9	5.7^{A}	0.6
	P	17	0.2	4.0	2.3^{B}	0.3
(b)	P1	3	0.5	2.1	1.4^{B}	0.5
	P2	3	3.3	4.0	3.6^{A}	0.2
	P3	3	3.0	3.8	3.3^{A}	0.3
	P4	3	2.9	3.9	3.5^{A}	0.3
	P5	3	0.2	0.7	0.5^{B}	0.2
	P6	2	1.1	1.2	1.1^{B}	0.1

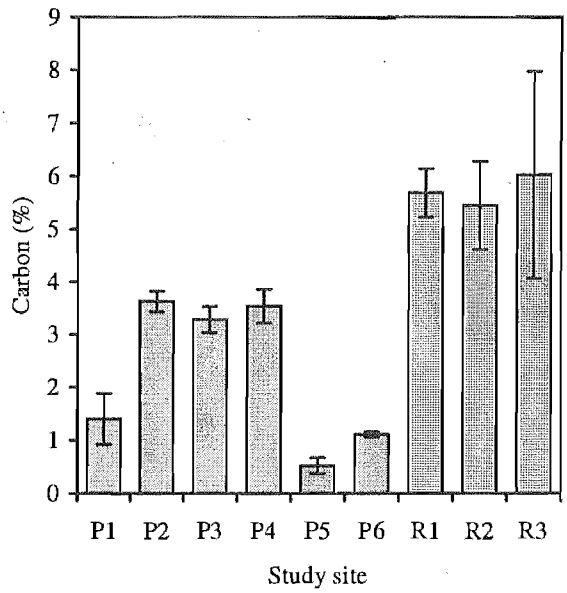


Figure 5.9: Mean percentage carbon of soil samples collected. The error bars depict the standard error around the mean.

Nitrogen

Total nitrogen levels varied significantly between remnant and planted restoration study sites (F = 11.77, df = 1, P = 0.002). This difference was the result of significantly higher mean nitrogen levels in remnant study sites than that of soil samples from planted restoration study sites. Similarly, nitrogen levels differed significantly between planted restoration study sites (F = 4.68, df = 5, P = 0.034). A Tukey's test was run to determine the nature of these differences. As with total carbon levels, soils of P2, P3 and P4 were shown to have significantly higher total

nitrogen than the P1, P5 and P6 study sites (Table 5.8). The lowest nitrogen values (%) overall, were found in soil samples from the P5 study site. The highest nitrogen values of 0.4% were found in soil samples from R3. The highest nitrogen values from planted restoration study sites were found in P2 (Figure 5.10).

Table 5.8: Descriptive statistics of total nitrogen values of soil samples collected. Part (a) investigates the difference in nitrogen levels between remnant and planted restoration study sites, while part (b) compares nitrogen values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.2	0.6	0.3^{A}	0.0
	P	17	0.0	0.3	0.2^{B}	0.0
(b)	P1	3	0.0	0.1	0.1^{B}	0.0
	P2	3	0.3	0.3	0.3^{A}	0.0
	P3	3	0.2	0.3	0.2^{A}	0.0
	P4	3	0.2	0.3	0.3^{A}	0.0
	P5	3	0.0	0.0	0.0^{B}	0.0
	P6	2	0.1	0.1	0.1^{B}	0.0

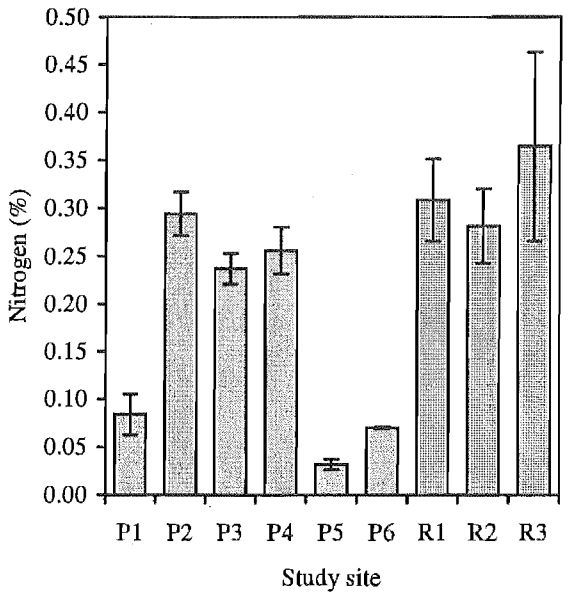


Figure 5.10: Mean percentage of nitrogen present in soil samples collected. Error bars are the standard error around the mean.

Olsen Phosphorous

Olsen phosphorous levels were not found to differ significantly between soil samples of remnant and planted restoration study sites (F = 1.35, df = 1, P = 0.256), nor was a significant difference detected between any of the six planted restoration study sites soil samples (F =2.39, df = 5, P = 0.106) (Table 5.9). The lowest Olsen phosphorous

value was found in soil samples collected from P3, and the greatest values were found in soil samples from P2 (Figure 5.11).

Table 5.9: Descriptive statistics of values of Olsen phosphorous for soil samples collected. Part (a) investigates the difference in Olsen P levels between remnant and planted restoration study sites, while part (b) compares Olsen P values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	4.0	14.6	10.0^{A}	1.3
	P	17	5.8	63.0	16.7^{A}	4.1
(b)	P1	3	6.0	8.6	6.9^{A}	0.8
	P2	3	21.7	63.0	37.4^{A}	12.9
	P3	3	5.8	7.8	7.1^{A}	0.7
	P4	3	10.3	53.4	26.0^{A}	13.8
	P5	3	8.7	15.1	12.2^{A}	1.9
	P6	2	7.2	7.2	7.2^{A}	0.0

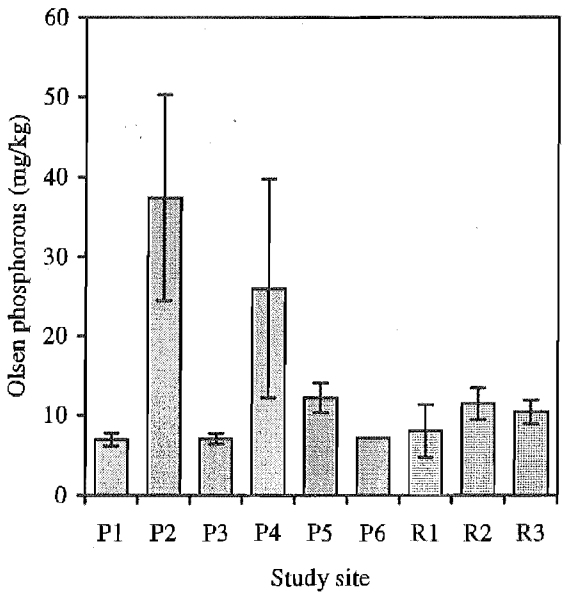


Figure 5.11: Mean levels of Olsen phosphorous present in soil samples collected. The error bars depict the standard error around the mean.

Cation Exchange Capacity

Planted restoration and remnant study sites were found to differ significantly in the mean cation exchange capacity of their respective soil samples ($F = 32.18$, df = 1, $P = <0.001$). The Tukey's test revealed that remnant study sites had significantly greater cation exchange capacity (CEC) in their soil than planted restoration study sites. An ANOVA run to detect differences in CEC between planted restoration study sites suggested that a significant difference was present ($F = 16.15$, df = 5, $P = <0.001$). According to the Tukey's test, P2 and P4 both had significantly higher cation

exchange capacity than P1, P5 and P6. P3 was also revealed to have significantly higher CEC than P5 and P6 (Table 5.10). The lowest mean CEC of all soil samples was found to be present in P5. P2 had the highest CEC of soil samples from planted restoration study sites (Figure 5.12), while R3 had the highest mean CEC of all study sites (18.7 cmol(+)/kg).

Table 5.10: Descriptive statistics of cation exchange capacity (cmol(+)/kg) values (CEC) present in soil samples. Part (a) investigates the difference in CEC levels between remnant and planted restoration study sites, while part (b) compares CEC values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	11.9	28.4	17.4^{A}	1.6
	P	17	2.0	12.9	8.0^{B}	0.9
(b)	P1	3	3.1	9.0	6.1^{B}	1.7
	P2	3	9.9	12.9	11.7^{A}	0.9
	P3	3	9.8	10.6	10.2^{AB}	0.2
	P4	3	10.4	10.8	10.7^{A}	0.1
	P5	3	2.0	3.5	3.0^{C}	0.5
	P6	2	5.2	5.2	5.2^{C}	0.0

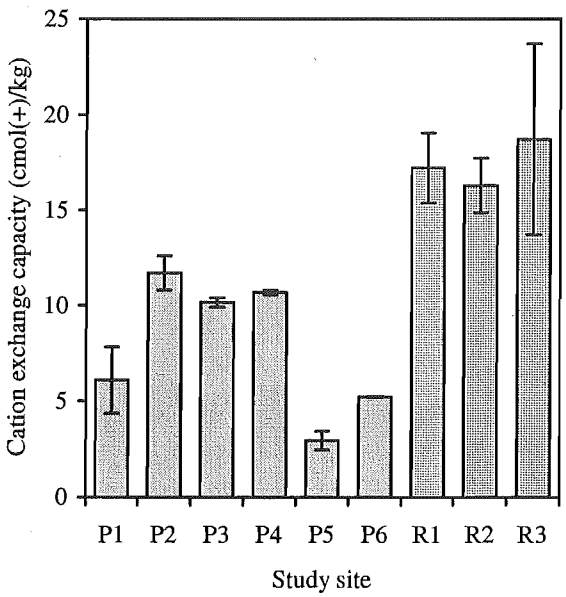


Figure 5.12: Mean cation exchange capacity of soil samples collected. The error bars represent the standard error around the mean.

Exchangeable Bases

Calcium

A significant difference was suggested by an ANOVA between soil calcium levels in soil samples of remnant and planted restoration study sites (F = 17.12, df = 1, P = < 0.001). According to the Tukey's test calcium levels were significantly higher

in remnant study sites than planted restoration study sites. Similarly, ANOVA suggested a significant difference between soil calcium levels in planted restoration study sites (F = 3.62, df = 5, P = 0.035). The Tukey's test revealed that P1 had significantly larger amounts of exchangeable calcium than P3 and P5 (Table 5.11). The lowest mean calcium levels were found in soil samples from P4, and the greatest calcium levels came from R3 (9.7 cmol(+)/kg). The highest mean calcium values of planted restoration study sites were found in P1 soil samples (Figure 5.13a).

Table 5.11: Descriptive statistics of exchangeable calcium (cmol(+)/kg) values present in soil samples. Part (a) investigates the difference in exchangeable calcium levels between remnant and planted restoration study sites, while part (b) compares exchangeable calcium values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	1.3	14.3	5.7^{A}	1.5
	P	17	0.2	5.3	1.1^{B}	0.3
(b)	P1	3	1.5	5.3	3.0^{A}	1.1
	P2	3	0.6	0.9	0.8^{AB}	0.1
	P3	3	0.2	1.1	0.5^{B}	0.3
	P4	3	0.6	0.8	0.7^{AB}	0.1
	P5	3	0.2	0.8	0.4^{B}	0.2
	P6	2	0.7	1.1	0.9^{AB}	0.2

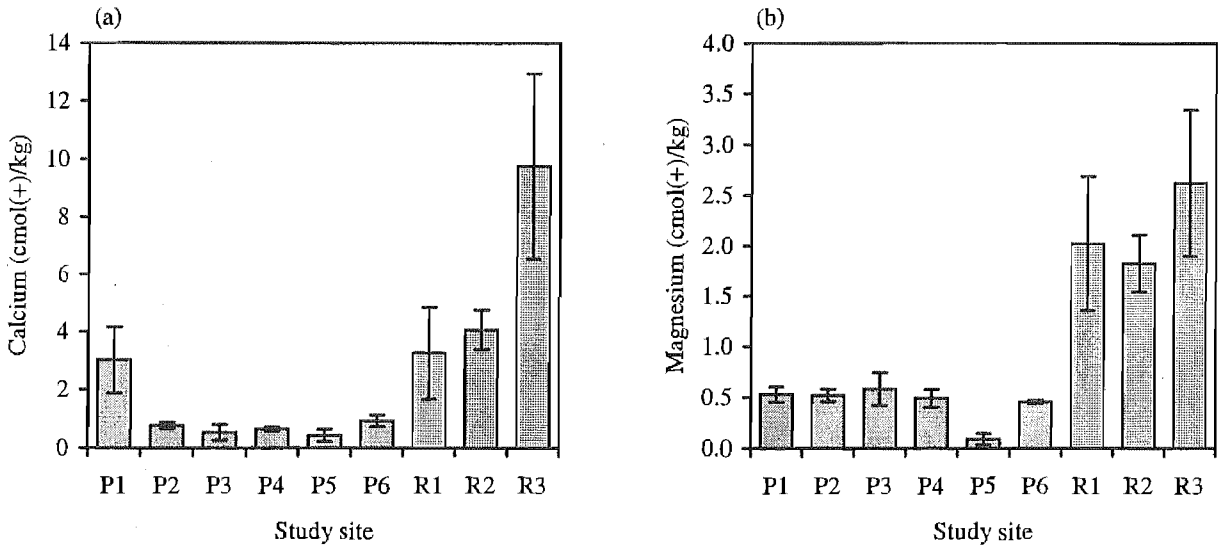


Figure 5.13: Graph (a) illustrates the mean levels of calcium present in soil samples collected from each of the nine study sites, while (b) shows mean exchangeable magnesium levels. The error bars, on both graphs, depict the standard error around the mean.

Magnesium

Levels of exchangeable magnesium differed significantly between planted restoration and remnant study sites (F = 51.53, df = 1, P = <0.001). Remnant study sites were shown to have significantly higher magnesium levels than planted restoration study sites using Tukey's 'comparison of means' test. An ANOVA suggested that magnesium levels also differed significantly between the six planted restoration study sites (F = 3.72, df = 5, P = 0.032). The Tukey's test revealed that soil samples from P5 contained significantly less exchangeable magnesium than P3 (Table 5.12). Soil samples from P5 contained the least amount of exchangeable magnesium of all study site soil samples. The highest exchangeable magnesium levels were detected in soil samples from R3 (2.6 cmol(+)/kg). The highest exchangeable magnesium levels within planted restoration study sites were found in soil samples from P3 (Figure 5.13b).

Table 5.12: Descriptive statistics of exchangeable magnesium (cmol(+)/kg) values present in soil samples. Part (a) investigates the difference in exchangeable magnesium levels between remnant and planted restoration study sites, while part (b) compares exchangeable magnesium values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	1.0	4.0	2.2^{A}	0.3
	P	17	0.0	0.9	0.4^{B}	0.1
(b)	P1	3	0.4	0.6	0.5^{AB}	0.1
	P2	3	0.4	0.6	0.5^{AB}	0.1
	P3	3	0.4	0.9	0.6^{A}	0.2
	P4	3	0.4	0.7	0.5^{AB}	0.1
	P5	3	0.0	0.2	0.1^{B}	0.1
	P6	2	0.4	0.5	0.5^{AB}	0.0

Potassium

An ANOVA run using exchangeable potassium levels by type of study area (remnant or planted restoration study sites) suggested the presence of a significant difference (F = 7.41, df = 1, P = 0.012). According to the Tukey's test, exchangeable potassium levels were significantly higher in soil samples from remnant study sites. An ANOVA also detected a significant difference in levels of exchangeable potassium between the six planted restoration sites (F = 5.54, df = 5, P = 0.009). According to the Tukey's test, P1 had significantly higher levels of exchangeable potassium in soil samples than P5 and P6 (Table 5.13). The highest potassium levels were found in soil samples from R3 (0.3 cmol(+)/kg), the highest levels within planted restoration study sites occurred

in P1. Soil samples from the P5 study site were found to contain the least amount of potassium overall (Figure 5.14a).

Table 5.13: Descriptive statistics of exchangeable potassium (cmol(+)/kg) values present in soil samples. Part (a) investigates the difference in exchangeable potassium levels between remnant and planted restoration study sites, while part (b) compares exchangeable potassium values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.1	0.6	0.2^{A}	0.1
	P	17	0.0	0.2	0.1^{B}	0.0
(b)	P1	3	0.1	0.2	0.2^{A}	0.0
	P2	3	0.1	0.1	0.1^{AB}	0.0
	P3	3	0.1	0.1	0.1^{AB}	0.0
	P4	3	0.1	0.2	0.1^{AB}	0.0
	P5	3	0.0	0.0	0.0^{B}	0.0
	P6	2	0.0	0.1	0.0^{B}	0.0

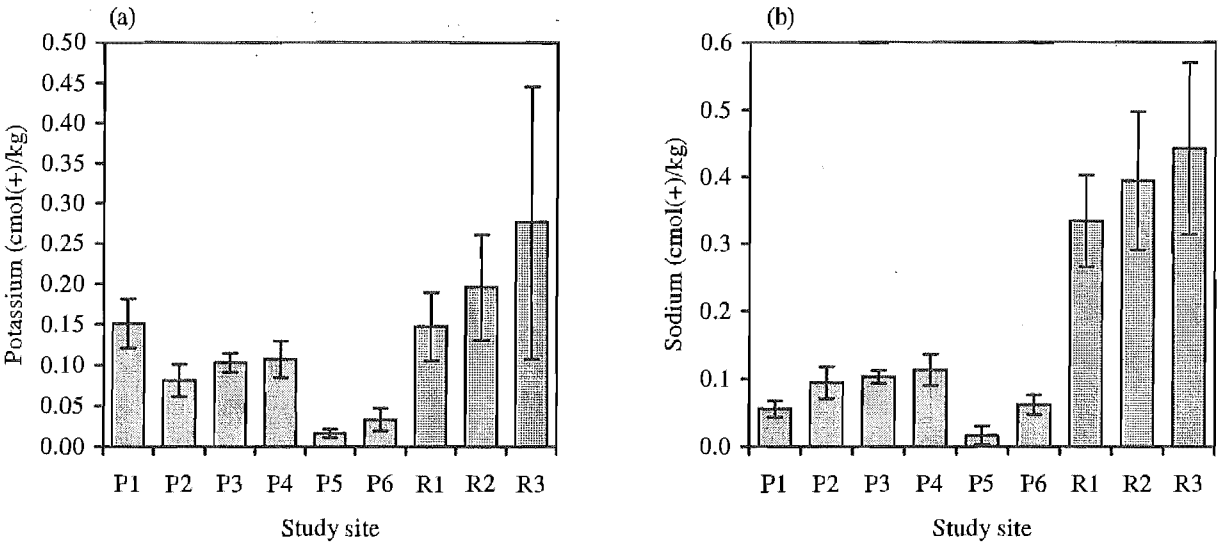


Figure 5.14: Graph (a) illustrates the mean levels of exchangeable potassium present in soil samples collected from each of the nine study sites, while (b) shows mean exchangeable sodium levels. The error bars, on both graphs, depict the standard error around the mean.

Sodium

A significant difference in exchangeable sodium levels between planted restoration and remnant study sites was detected by an ANOVA ($F = 59.62$, df = 1, $P = <0.001$). Tukey's test revealed that soil samples from remnant study sites had significantly higher levels of exchangeable sodium than those from planted restoration study sites. A further significant difference in exchangeable sodium levels was suggested between planted restoration study sites ($F = 4.61$, df = 5, $P = 0.016$). According to the Tukey's

test, P5 had significantly lower sodium levels than the P3 and P4 study sites (Table 5.14). Soil samples from P5 were found to contain the least amount of sodium of all study sites. The highest levels of sodium within soil samples tested were found within the R3 study site (0.4 cmol (+)/kg); the highest sodium levels within planted restoration study sites were detected in the P4 study site (Figure 5.14b).

Table 5.14: Descriptive statistics of exchangeable sodium (cmol(+)/kg) values present in soil samples. Part (a) investigates the difference in exchangeable sodium levels between remnant and planted restoration study sites, while part (b) compares exchangeable sodium values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	0.2	0.7	0.4^{A}	0.1
	P	17	0.0	0.2	0.1^{B}	0.0
(b)	P1	3	0.0	0.1	0.1^{AB}	0.0
	P2	3	0.1	0.1	0.1^{AB}	0.0
	P3	3	0.1	0.1	0.1^{A}	0.0
	P4	3	0.1	0.2	0.1^{A}	0.0
	P5	3	0.0	0.0	0.0^{B}	0.0
	P6	2	0.1	0.1	0.1^{AB}	0.0

Base Saturation

An ANOVA suggested the presence of a significant difference in base saturation levels between planted restoration study sites ($F = 7.05$, df = 1, $P = 0.014$). Remnant study sites were shown, using Tukey's 'comparison of means' test, to have significantly greater base saturation levels than planted restoration study sites. Similarly, a significant difference in base saturation levels was detected, using ANOVA, between the six planted restoration study sites ($F = 17.01$, df = 5, $P = <0.001$). The Tukey's test revealed that base saturation levels of soil samples from P1 were significantly higher than all other planted restoration study sites (Table 5.15). The base saturation levels at this study site were higher than those of soil samples from all other study sites. The lowest mean base saturation levels were shared by P2 and P4 soil samples (Figure 5.15).

Table 5.15: Descriptive statistics of base saturation (%) values present in soil samples. Part (a) investigates the difference in base saturation values between remnant and planted restoration study sites, while part (b) compares base saturation values between the six planted restoration sites. The superscript letter indicates the Tukey grouping; means with the same letter were not significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	9	17.3	88.2	46.7^{A}	7.3
	P	17	8.5	68.2	24.0^{B}	4.8
(b)	P1	3	49.1	68.2	61.7^{A}	6.3
	P2	3	8.9	16.8	12.7^{B}	2.3
	P3	3	8.5	20.7	12.8^{B}	4.0
	P4	3	10.9	16.0	12.7^{B}	1.7
	P5	3	9.6	31.6	17.4^{B}	7.1
	P6	2	25.5	31.2	28.4^{B}	2.8

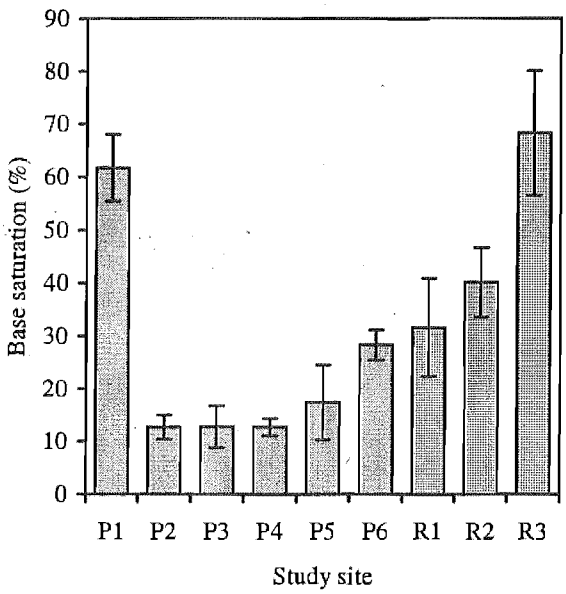


Figure 5.15: Mean base saturation levels of soil samples collected. The error bars depict the standard error around the mean.

5.3.3. Seed Rain

A total of 1,602 seeds from at least twelve species were found within the twenty six seed traps over the five month collection period. A substantial number of these were identified as seeds of exotic grasses that dominate the ground cover within planted restoration study sites. However, a total of 1,117 seeds of woody species were caught in seed traps established throughout the study area. Table 5.16 details the number of seeds caught within study sites. Within planted restoration study sites, 56.7% seeds caught were from woody species, all of which had bird dispersal as their primary mode of dispersal. The remaining seeds collected in the planted restoration study sites

were those of exotic grasses. All seeds collected in remnant study sites were of woody species; two of a total of 484 seeds were wind dispersed seeds.

Table 5.16: Descriptive statistics of the number of seeds found within seed traps. Part (a) compares the number of seeds found in planted restoration and remnant study sites. Part (b) details the number of seeds found in planted restoration study sites.

		Number of Seed Traps	Minimum	Maximum	Mean	Standard Error
(a)	P	17	0.0	239.0	65.7	17.1
	R	9	0.0	263.0	53.9	29.4
(b)	P1	3	1.0	64.0	24.3	19.9
	P2	3	13.0	163.0	67.3	48.0
	P3	3	13.0	117.0	66.7	30.1
	P4	3	36.0	149.0	75.3	36.9
	P5	3	2.0	239.0	126.7	68.7
	P6	2	0.0	36.0	18.0	18.0

The dominant dispersal mode of seeds caught within seed traps throughout the study area was bird dispersal (Table 5.17). Only three wind dispersed species were located throughout the study area. *Holcus lanatus* was particularly prevalent throughout the planted restoration sites, which was unsurprising given that seed traps were frequently located amidst the seedling grass species.

Table 5.17: Dispersal mode of seed rain caught within planted restoration and remnant study sites. The symbol √ indicates the presence of a particular species in either of the two types of study site.

Species	Dispersal Mechanism	Planted Study Restoration Sites	Remnant Study Sites
Coprosma grandifolia	bird	√	√
Coprosma propinqua	bird	√	
Coprosma robusta	bird	√	√
Dacrycarpus cupressinum	bird		√
Hedycarya arboreus	bird		√
Myrsine salicina	bird	√	
Pittosporum crassifolium	bird	√	
Pittosporum eugenioides	bird	√	
Pseudopanax crassifolius	bird	√	
Celmisia spp.	wind	√	
Holcus lanatus	wind	√	
Parsonsia heterophylla	wind		√

Species richness of seed rain ranged from zero to four species per seed trap. An ANOVA run to determine whether a significant difference existed in species richness of all species collected in seed rain between planted restoration and remnant study sites revealed the absence of such a difference ($F = 0.96$, df = 1, $P = 0.337$). Similarly,

no significant difference in the number of species contained in seed traps was found between the six planted restoration study sites ($F = 1.26$, df = 5, $P = 0.348$).

A further ANOVA was run to investigate whether a significant difference was present in the species richness of seed rain between different study sites when only woody species were used. A significant difference was found to be present between planted restoration and remnant study sites ($F = 6.41$, df = 1, $P = 0.018$). A Tukey's test was run to determine the nature of this difference. This 'comparison of means' test revealed that remnant study sites had significantly more woody species present in the seed traps than planted restoration study sites. No significant difference was apparent in the species richness of woody species within seed rain collected in planted restoration study sites ($F = 2.51$, df = 5, $P = 0.095$).

5.3.4. Light (Visible Sky)

The amount of visible sky apparent in study sites was tested using an ANOVA between the two types of study sites. This test suggested that the amount of visible sky present in planted restoration sites differed significantly from remnant study sites (F = 100.49, df = 1, P = <0.001). The Tukey's test was used to elaborate on this significant difference, revealing that significantly more sky was visible in planted restoration study sites than remnant study sites. Additionally, a significant difference was detected by ANOVA in the amount of visible sky in the six planted restoration study sites (F = 30.47, df = 5, P = <0.001). According to the Tukey's test, more sky was visible in P1 than all other planted restoration study sites. P2 and P4 had significantly more sky visible than P5 and P6, while P2 also was found to have significantly more visible sky than the P3 study site (Table 5.18; Figure 5.16).

Table 5.18: Descriptive statistics of the proportion of visible sky present in study sites. Part (a) compares visible sky present in remnant and planted restoration study sites, while part (b) denotes descriptive statistics for values of visible sky for each of the six planted restoration study sites. Superscript letters indicate the Tukey grouping; means with the same letter were not found to be significantly different.

		n	Minimum	Maximum	Mean	Standard Error
(a)	R	45	0.0	0.1	0.1^{A}	0.0
	P	85	0.0	0.9	0.4^{B}	0.0
(b)	P1	15	0.6	0.9	0.8^{A}	0.0
	P2	15	0.3	0.8	0.5^{B}	0.0
	P3	15	0.2	0.7	0.3^{C}	0.1
	P4	15	0.2	0.7	0.4^{BC}	0.1
	P5	15	0.0	0.4	0.3^{D}	0.0
	P6	10	0.2	0.4	0.2^{D}	0.0

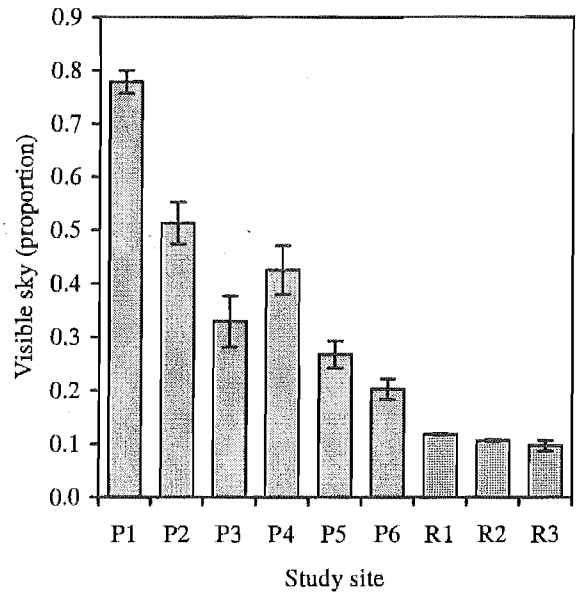


Figure 5.16: Mean proportion of visible sky present in each of the nine study sites. The error bars depict the standard error around the mean.

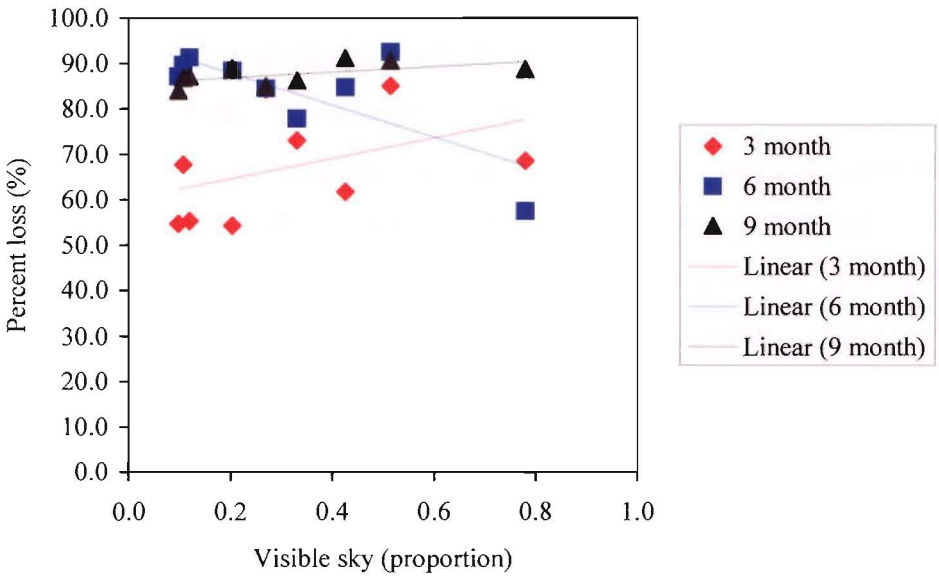


Figure 5.17: Comparison of litter decomposition and visible sky.

Figure 5.17 illustrates the relationship between visible sky and litter decomposition (Section 5.3.1.). There was a tendency for the amount of mass lost due to litter decomposition to increase with increasing light levels (i.e. proportion of visible sky) for measurements obtained after litter bags had been left in the field for three months. The same trend occurred, less dramatically for litter bags that had been in the field for nine months. The opposite trend occurred, however in the six month litter decomposition litter bags. However, this trend was primarily due to the low level of litter decomposition recorded for litter bags in the P1 study site.

5.4. Preliminary Discussion of Results

The discussion of results in this chapter is sectionalised and brief, to allow for in-depth interpretation and integration of the results in Chapter 6.

5.4.1. Ground Litter

Litter dynamics represent an important section of nutrient cycling and energy transfer in forest ecosystems. Consideration of litter dynamics was regarded as an essential aspect of determining the level of success of restoration at Cape Foulwind, as the growth and productivity of forest ecosystems is dependent on the nature of and rate of decomposition of forest litter (Kavvadias et al, 2001).

Litter decomposition is an essential aspect of nutrient cycling and the productivity of forests (Didham, 1998). The decomposition of forest litter is the major pathway for provision of organic and inorganic elements to nutrient cycling processes (Kavvadias et al, 2001; Belyea, 1996). Three primary factors are influential in the litter decomposition: litter quality, abiotic conditions (e.g., moisture, pH, temperature, oxygenation), and the nature and abundance of decomposing organisms (Belyea, 1996). It is also affected by climate on a regional scale (Belyea, 1996), aspect, slope, litter supply, species, abundance of understory vegetation, and soil fertility (Kavvadias et al, 2001). Blair et al (1990) suggest that resource quality may be the chief factor affecting invertebrate abundances with microclimatic conditions secondary. By affecting the abundance, composition and activity of decomposer organisms, resource quality is a major factor controlling rates of organic matter decomposition and nutrient release in forest ecosystems (Blair et al, 1990).

The nature and abundance of decomposer organisms and their interaction with fauna are also relevant to litter decomposition (Cortez, 1998). Decomposition is primarily accomplished by microorganisms, but invertebrates play an important stimulatory role (Sulkava & Huhta, 1998). Ground active invertebrates have important direct and indirect effects on litter decomposition rates (Didham, 1998).

Litter bags have several limitations as a means of determining litter decomposition (Hector et al, 2000). Litter decomposition was assessed in terms of litter mass remaining at the end of the three experimental periods. This approach limited the amount of information that could be obtained from this aspect of the study. For example, changes in the rate of decomposition could not be determined as all bags started with approximately 2g of litter and would eventually empty, differences in the time taken to get to that point could not be known (Hector et al, 2000). The use of *Coprosma robusta* leaves as a proxy for determining litter decomposition in each of the nine study sites, was justified as the purpose of this aspect of the study was to investigate the affect of microclimate on litter decomposition, rather than to determine the affect of species diversity, or varying decomposition rates of species present in the study area. The amount of litter placed within the bags was relatively small, possibly

restricting the extent to which differences in decomposition could be detected (Hector et al, 2000).

Only six replicates were used within each study site for each of the three time periods. This level of replication is small; therefore results must be interpreted with caution. Also due to practical constraints, the length of time that litterbag trials were run was relatively short. It is important that this is taken into account when considering the conclusions drawn from this study at Cape Foulwind. Care has been taken not to extrapolate these short-term results to infer trends of long-term litter decomposition, as decomposition rates tend to decline with time (Berg, 2000; Hector, et al, 2000; Blair et al, 1990).

With the exception of monoculture plantations, litter of more than one tree species is usually mixed on the forest floor. This mixing of litter from different species and different resource quality may affect decay rates and nutrient fluxes within forest ecosystems (Blair et al, 1990). The lack of mixing of litter of different species within this experiment means that natural conditions were not mimicked. However, as the aim of this portion of the study was to determine the affect of microclimatic conditions, and not the affect of the chemical composition of species (Berg, 2000), species richness or interaction of different species on litter decomposition, this deviation from natural conditions was acceptable. The effect of litter quality and microhabitat were found to be highly significant on mass loss by Belyea (1996).

Results obtained within this litter decomposition aspect of the study may have been affected by processes other than decomposition, such as erosion influx of fine particles, exclusion of decomposer organisms larger than the mesh size, and colonisation of the experimental material by roots, fungi, bacteria and invertebrates (Belyea, 1996).

5.4.1.1. Did Ground Litter Vary Between Planted Restoration and Remnant Study Sites?

Remnant study sites had significantly deeper and greater volume of ground litter than did the planted restoration study sites. The development of ground litter is an essential aspect of planted restoration study sites progressing towards the benchmark remnant study sites.

Regeneration density was found to increase with increasing litter depth / volume, which may be attributed to the development of appropriate microclimatic conditions with increasing litter depth. However, this result is contrary to that found by Parrotta (1995) in a Northern Hemisphere study, which found negative correlations between seedling density and both litter depth and dry mass.

No significant difference was apparent in litter decomposition in planted restoration and remnant study sites, despite the general tendency for litter decomposition to occur at greater levels in planted restoration than remnant study sites (Figure 5.7). Initial loss of mass within litter bags occurred quickly throughout all study sites. The greatest loss occurred within the initial three month period in the field; all litter bags lost more than 50% of their original estimated dry weight.

5.4.1.2. Did Ground Litter Vary Between Planted Restoration Study Sites?

A clear progression of increasing litter depth and volume was apparent. Both litter depth and litter volume increased towards remnant study sites. The greatest amount of litter (both depth and litter) was found in the P6 study site. Although significantly more volume of litter than all other study sites in planted restoration sites was found in P6, the litter layer in this study site was not found to be significantly deeper than the P3 study site. No evidence of a litter layer was apparent in the 'youngest' study site (P1). Litter depth is affected by moisture and stage of decomposition.

Numerous samples of ground litter measurements (both depth and volume) were taken. Accordingly, a reliable indication of the status of ground litter in each of the nine study sites was expected. Only ground litter that resulted from natural death was counted. In other words, dead material resulting from the application of herbicide

spray in study sites was not included in this study. Some high litter depth values within planted restoration study sites were the result of litter not lying directly on the ground surface, and are not necessarily indicative of the actual amount of litter. However, the subsequent collection of litter to obtain volume measurements would have accounted for this.

Within planted restoration study sites, ground litter was only observed directly under trees where the invasive grass sward was suppressed. This serves as further evidence that the establishment of a canopy cover is essential for the development of a self-sufficient fully functioning ecosystem.

Furthermore, a significant difference was not apparent in the percent mass loss that occurred within litter bags placed in study plots in planted restoration study sites. For the first two periods (three and six months) in the field the greatest amount of litter decomposition occurred in the P2 study site. Because no significant difference was found between the amount of litter lost between any of the study sites it can tentatively be inferred that microclimatic conditions did not vary significantly between the six planted restoration study sites.

5.4.2. Soil

Soil is the starting point for plants; its properties are crucial to the degree to which a community can develop at a site (Bradshaw, 1987). Through their physical properties and biological activities, soils form an environmental factor influential in the formation and control of the structure and function of the ecosystem they support. Soil attributes are fundamental to the success of ecological restoration (Marrs, 2002; Ross et al, 1995). Soil provides four basic functions vital for plant growth: the supply of water, nutrients and air (gaseous exchange), while providing physical support affecting plant growth (Ross et al, 1995). Soil organic matter (SOM) is a major source of plant nutrients. It also affects physical conditions such as water retention, soil structure and aeration (Ross et al, 1995); however, the accumulation of SOM is an extremely slow process (Berg, 2000).

Biological processes of nutrient accumulation are important for the development of a fully functioning ecosystem (Dobson et al, 1997; Webb, 1996). If nutrients are not

available in adequate quantities in the soil, ecosystem development will be restricted (Bradshaw, 1983). The supply of nutrients to plants is mediated by soil microorganisms and fauna through their impact on the decomposition of dead organic matter and nutrient mineralisation (Marrs, 2002; Brussaard et al, 1996). Soil organisms also influence plant nutrition directly through interactions with the root system (Brussaard et al, 1996; Marrs, 2002). The diversity of soil-bourne flora and fauna is as important as the diversity of above ground organisms (Ross et al, 1995). The importance of microorganisms to ecological restoration is covered in detail by Allen et al (2002). Detailed investigation of soil microorganisms, despite its obvious importance to the success of restoration, was beyond the scope of this research project.

The primary reasoning behind, and value of, soil measurements undertaken during this study was to establish a set of values to be used for future comparisons. Such comparisons will enable the future determination of whether a progression in nutrient status from that which currently exists in planted restoration sites towards that present in soil of remnant study sites, which are seen as a benchmark for restoration success, has occurred. The number of soil samples able to be analysed for this study was limited due to financial constraints. However, results are directly comparable between study sites because the same collection methods were used throughout.

5.4.2.1. Did Soil Attributes Vary Between Planted Restoration and Remnant Study Sites?

Soil nutrient availability changes in response to pH. The leaching of many elements is increased in acidic conditions (Marrs, 2002). Plants take up cations and release protons into soil, a process compensated by organic matter mineralisation. The rate of production of organic matter is initially higher than the rate of organic matter mineralisation, due to the rapid growth of vegetation; this leads to soil acidification (Honnay et al, 2002). Honnay et al (2002) suggest that accumulation of soil organic matter, resulting from the slowed decay processes, will only commence once soil pH has become sufficiently low. All soil samples collected from study sites were acidic. No significant difference was found between the level of acidity in soil samples from planted restoration and remnant study sites.

There was no significant difference detected by ANOVA in the level of Olsen phosphorous found in soil samples from either type of study sites. In all other tests, however, soil samples from remnant study sites were found to contain significantly higher quantities of nutrients. That is, total carbon, total nitrogen, and potassium levels were all significantly higher in remnant study sites.

Cation exchange capacity was found to be significantly higher in remnant study sites than planted restoration study sites. The cation exchange capacity of a soil is a quantitative measure of the soil's ability to hold exchangeable cations (McLaren & Cameron, 1997). Mean values of cation exchange capacity of soil samples from both remnant and planted restoration study sites fell within the range of typical cation exchange values reported by McLaren & Cameron (1997) of between 5 and 30 cmol(+)/kg. Exchange sites in soils are dominated by the exchangeable bases Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}, levels of which were tested during this study, and acidic cations (H^{+} and Al^{3+}) (McLaren & Cameron, 1997). Exchangeable bases are considered to be available for plant uptake. Values of all four exchangeable bases were found to be significantly higher in soil samples from remnant study sites than soils in planted restoration study sites. The usual order of abundance of these exchangeable bases is normally $Ca^{2+} > Mg^{2+} > K^{+} > Na^{+}$, with calcium being by far the dominant cation (McLaren & Cameron, 1997). This was indeed the order, with respect to dominance, that the cations were found within planted restoration study sites, and primarily in remnant study sites. The exception was that the mean value of Na^{+}, in remnant study sites, was greater than the mean value of K^{+}.

The sum of the four exchangeable bases referred to above in the soil is referred to as the total exchangeable bases. The proportion of the cation exchange capacity occupied by these bases is referred to as base saturation (McLaren & Cameron, 1997). Base saturation levels of soil samples from remnant study sites were also found to be significantly higher than samples from planted restoration study sites.

5.4.2.2. Did Soil Attributes Vary Between Planted Restoration Study Sites?

Significant differences were found for all tests undertaken on soils when investigating differences between samples planted restoration study sites, with the exception of Olsen phosphorous; levels of which did not differ significantly between any of the six study sites. P2, P3 and P4 were found to have significantly higher levels of total carbon and nitrogen than soil samples from P1, P5, and P6. In both cases, P5 had the lowest levels of these nutrients, while P1 contained the highest percentage of carbon and soil samples from the P2 study site was found to have the highest levels of nitrogen within planted restoration study sites. For all study sites however, R1 and R3 soil samples had the highest amount of carbon and nitrogen, respectively. P5 soil samples had the lowest levels of potassium, also, while P1 and R3 had the highest.

Nitrogen accumulation may be restricted by phosphorous deficiency (Marrs, 2002). However, as nitrogen and phosphorus were supplied together at planting as NPK fertiliser, phosphorus deficiency is unlikely to have played an influential role in controlling nitrogen levels. Potassium levels were found to be significantly higher in P1 than in soil samples from P6. The obvious lack of nutrients in P5 and P6 soils may be attributed to the time since planting. A two year slow release fertiliser pellet (Keir, 1998) was placed in each hole that was planted into. Thus, since the longest periods of time have lapsed since planting was undertaken in the P5 and P6 study sites, fewer nutrients are available in the soil. Although litter layer has developed in P6, one is lacking in the P5 study site, therefore time is required to build up SOM that will provide nutrients.

Nitrogen is a commonly limiting factor for plant growth. It is the only nutrient that changes significantly with ecosystem development and that can be shown to be continually limiting (Bradshaw, 1987). There are only two ways that nitrogen accumulation can be accelerated, through the addition of fertiliser and through biological fixation. Species with symbiotic nitrogen-fixing organisms, *Trifolium repens* and *Ulex europaeus*, are present in planted restoration study sites (Marrs, 2002). These species should provide a longer lasting solution to nitrogen deficiency until a sufficient soil organic matter content has been developed in these study sites. Nitrogen inputs have important direct effects on the supply of nitrogen to vegetation.

Bakker & Berendse (1999) found that indirect effects resulting from the accumulation of SOM lead to an accelerated increase in nitrogen mineralisation.

Cation exchange capacity of a soil, a quantitative measure of the soil's ability to hold exchangeable cations (McLaren & Cameron, 1997), was determined to be significantly higher in soil samples from P2 and P4 than those from P1, P5 and P6. P3 soil samples were also found to have greater CEC than those from P5 and P6. Cation exchange capacity was suggested by Aronson et al (1993a) as necessary to include as a vital functional ecosystem attribute due to its universal applicability, sensitivity to degradation, and because it is directly correlated with overall soil fertility. Cation exchange values of soil samples of the P5 study site fell below the range of typical values for New Zealand soils.

A significant difference was suggested by ANOVA between mean values of the four exchangeable bases tested in this study, Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}, in planted restoration study sites. P1, P3 and P4 soil samples were found to contain significantly higher levels of some exchangeable bases than P5. Soil samples from P5 contained the lowest levels of all four exchangeable bases, except for Ca^{2+}, for which P4 study site samples contained the lowest amount. Ca^{2+} was found in the highest concentration in all planted restoration study site soil samples except P3, for which Mg^{2+} was found in greater quantities. Levels of K^{+} and Na^{+} were similar, if not equal, between each study site. In soil samples from P2 and P6, Na^{+} was found in greater quantities than K^{+}. Base saturation levels were significantly higher in P1 than in soil samples from all other planted restoration study sites. This may be due to the relatively recent application of fertiliser, at the time of planting.

The importance of soil in ecological restoration programmes cannot be overstated. In order to achieve successful restoration of this aspect of the ecosystem both the starting point and desired target must be known (Marrs, 2002). Development of soil organic matter with a significant nitrogen content is necessary. As the soil organic matter develops there is a need to ensure that decomposition is occurring so that nutrients are available to further ecosystem development (Marrs, 2002). Litter decomposition is an aspect of ecosystem functioning that is discussed in section 5.4.3.2.

5.4.3. Seed Rain

Consideration of seed rain is implicit to achieving successful regeneration (Dungan et al, 2001). Regeneration of woody species is dependent on propagules being dispersed to a site where they are able to germinate and grow into seedlings (Honnay et al, 2002; Bakker & Berendse, 1999). Dispersal may occur via abiotic means such as wind, gravity or water, or with the aid or biotic vectors such as birds or insects. A combination of these means may also result in seed dispersal. Failure to recognise the importance of seed dispersal may lead to the obstruction of patterns or processes that have important consequences for the direction of vegetation change and development in a community (Dungan et al, 2002).

A seed type may fail to arrive at a site due to a lack of a dispersal agent, or upon arrival may fail to germinate due to some other limiting factors, such as inappropriate microhabitat conditions (Honnay et al, 2002; Dungan et al, 2001). Species with narrow regeneration niches often require special 'safe sites' for germination and establishment (Cornett et al, 2001). Safe sites result from combinations of canopy dominance and microhabitat quality (Cornett et al, 2001). Habitat quality is influenced primarily by drainage litter composition and soil nutrient status (Honnay et al, 2002).

5.4.3.1. Did Seed Rain Vary Between Planted Restoration and Remnant Study Sites?

Almost half (43.3%) of the seeds obtained in seed traps in planted restoration study sites were seeds of exotic grasses (Section 5.3.3.), demonstrating an aspect of the pervasive nature of exotic grasses. No significant difference was detected between the number of different species caught in seed traps in planted restoration and remnant study sites when exotic grass species were included. However, a significant difference was suggested, when the ANOVA was limited to investigating the number of woody species. Remnant study sites were found to have significantly more woody species dispersed into seed traps than did planted restoration study sites. This difference may be the result of the vegetation structure in each type of study sites. Because remnant study sites have an extensive canopy cover, all seed traps were directly below the canopy. Seeds may have dropped directly into traps via gravitational dispersal, or, given the canopy structure, may have been below perch sites for bird species.

Deposition of bird dispersed seeds has been found to be limited by the availability of suitable perch sites (Ferguson & Drake, 1999). The very high numbers of seeds that were present in some seed traps at Cape Foulwind may have been due to preferential perching by birds at particular locations. Alternatively, the positioning of the seed traps near to or directly under some species may have resulted in large numbers of seeds falling directly from trees into the seed traps.

5.4.3.2. Did Seed Rain Vary Between Planted Restoration Study Sites?

No significant difference was detected by ANOVA in the number of woody species found in seed traps within planted restoration study sites. A large degree of variation was present in the number of individual seeds caught within each study site. Similar to results of Ferguson & Drake's (1999) study on Mana Island, New Zealand, most bird dispersed seeds were apparently being deposited below existing vegetation. Only two of the 12 species dispersed into planted restoration study sites were of species not currently growing in these sites. Regenerating seedlings appearing in planted restoration sites may provide more conclusive evidence of the importance of bird dispersal in facilitating establishment of woody species not extant in restoration sites (Table 5.17).

All woody species caught within seed traps, except *Pittosporum crassifolium*, were also identified as regenerating seedlings within study sites at Cape Foulwind. Seven different species of seed were detected within planted restoration study sites, leading to the inference that dispersal into these study sites is likely to result in successful germination (and potential establishment) of these species.

Results of seed rain may merely be a consequence of the relative paucity of traps in the study area. Dispersal limitation plays an important role for forest species and is affected by the degree of connectivity between sites (Honnay et al, 2002). Seeds of some species may be underrepresented in the seed rain data for numerous reasons. Seeds of some species, although dispersed into planted restoration study sites, may have failed to land in a seed trap or small seeds may have passed through the shade cloth material, or may have been light enough to have been subsequently blown out of the traps. The number of seed traps established (one per 10 × 10 m study plot) was

relatively few, therefore the chances of catching a large proportion of the number of seeds dispersed were greatly diminished.

Although seed rain was sampled for approximately five months, the period chosen may not have coincided with the seeding of some species (Dungan et al, 2001). However, the sampling period used was chosen to coincide with the peak fruiting times of the majority of species present. For example, April is the time of peak availability of kahikatea fruit, although it is a species that fruits irregularly; in some year producing almost no fruit, while other years producing a plentiful crop (Robertson & Hackwell, 1995). Availability of fruit varies seasonally, with peak abundance and diversity occurring from late summer through to late winter (Lee et al, 1991). The aim of this portion of the study was not to gain a comprehensive assessment of seed rain, but rather to provide a comparison between the relative densities of seeds falling in each of the nine study sites. Therefore, such a representative sample of seed rain was deemed to be sufficient for the purposes of this research (compared with Ferguson & Drake (1999)).

Seed predation may have occurred on seeds once caught in seed traps. Predation of seeds and fruit in seed traps by invertebrate species was highly likely. There was also a considerable possibility of predation by possums (Dungan et al, 2001). The effect of seed predation on the results obtained was not quantifiable. However, the likelihood of seed predation was assumed to be equivalent in all study sites.

Human error involved in the transfer of seeds from the seed traps to the laboratory bench where seeds were counted may have lessened the number of seeds found. However, care was taken to ensure that all visible seeds were accounted for. Identification errors may have resulted from a paucity of knowledge of seed identification. However, through consultation with Webb & Simpson (2001), and a detailed knowledge of species present in the study area, identification was undertaken with some confidence.

The rate at which species were dispersed into different study sites cannot be determined as the seed traps were cleared only once, at the cessation of the five month

trapping period. It may have been better to consider only bird disseminated seed rain as this would have provided a measure of dispersal, and would have provided more information regarding the perching preferences of birds. Bird disseminated seed was not differentiated from seed which was dispersed by other means (wind, gravity) for the purposes of this study. Because the seed traps were emptied only once it would have been impossible to differentiate between seeds which had been dispersed into a seed trap bare from those that arrived surrounded by their fruit and subsequently dried and broken open during the time lapsed between trapping and counting (Dungan et al, 2001). Dispersal activities of birds have been inferred through the appearance of regenerating seedlings within study sites (see Section 3.3.3.).

The important role that birds play in the dispersal of seeds is well documented throughout the literature (McDonnell & Stiles, 1983). Vegetation structure and fruit availability can influence the perching behaviour of frugivorous birds and consequentially dispersal patterns (Ferguson & Drake, 1999; McDonnell & Stiles, 1983). The heterogeneous pattern of bird dispersal results in the frequent observation of high densities of bird disseminated seeds below the canopy of isolated trees or bushes (Debussche et al, 1982), with comparatively few seeds encountered in adjacent areas without perches (Ferguson & Drake, 1999). Likewise, the feeding habits of frugivorous dispersers will affect the nature of dispersal into the planted restoration study sites. The attraction that planted species have for birds, indicates that their presence encourages the arrival of novel species (Debussche et al, 1982). A constant source of fruiting and flowering plants was available to act as an attractant to dispersers into these restoration areas (McDonnell & Stiles, 1983). This was aided particularly by the long fruiting period of *Coprosma robusta*, from December 2001 through until September, 2002 (personal observation).

Such spatial variation in bird dispersal is particularly important in early successional areas, such as the planted restoration study sites where isolated trees can act as 'recruitment foci' (Ferguson & Drake, 1999). McDonnell & Stiles (1983) found that the presence of recruitment foci could increase seed input by more than an order of magnitude around the focus. The post-foraging behaviour of birds is an important component of their dispersal efficiency (Williams & Karl, 1996). Frugivorous birds

may influence vegetation patterns through of the species they disperse, but existing vegetation, particularly the presence of recruitment foci influence the recruitment of new species by affecting bird movement and ensuing dispersal patterns (McDonnell & Stiles, 1983). Birds are attracted to trees and shrubs, which at a minimum provide perching sites. This is an exponential process, with positive feedback between increasing density of woody species and increasing disperser visits (Robinson & Handel, 1993).

5.4.4. Light (Visible Sky)

Significantly more sky was visible in planted restoration study sites than remnant study sites. A general progression for amount of visible sky to decrease with increasing time since planting was apparent. Significantly more sky was visible in P1 than all other planted restoration study sites. Vegetation structure is obviously influential in determining the amount of light that reaches ground level. The herb and shrub stratum substantially modifies light levels. George & Bazzaz (1999a) report that ferns reduce light levels below their canopies to 32% of the already low light levels existing below the overstorey canopy. Further, light levels below the litter layer in remnant study sites decrease exponentially, substantially affecting microclimatic conditions for regeneration (George & Bazzaz, 1999a). The regeneration of seedlings and their successful growth to form part of the canopy depends primarily on light levels permitted by the absence of a complete canopy cover in planted restoration study sites (Wells et al, 1998).

The amount of visible sky may play a role in the rate of litter decomposition in this study undertaken at Cape Foulwind. In a comparison of light levels and litter decomposition (Figure 5.17), it was revealed that the amount of short-term litter decomposition increased with increasing amounts of visible sky. This pattern was evident in litter samples that had been left in the field for three and nine months. The opposite was shown by the six month litter bags, however numerous complicating factors may have influenced this result. Principally, this trend was due to the comparatively low amount of litter decomposition occurring in litter bags left in the P1 study site. P1 is exposed to strong onshore winds, which may have affected the rate of litter decomposition. Further, all other study sites, although lacking a dense

litter layer, litter bags were surrounded by an extensive grass sward. Such habitat features, although not tested specifically, would have affected microclimatic conditions resulting in altered decomposition rates.

5.5. Summary

Litter decomposition is occurring within all study sites. The occurrence of such an ecosystem process is valuable as decomposition is an essential component of nutrient cycling. The vegetation present in planted restoration study sites appears sufficient to attract bird species to disperse novel species into these areas, thereby facilitating regeneration of woody species and accelerating ecological succession of these plantings. Dispersal, although an integral part of regeneration, does not guarantee the establishment of desired species until soil and microclimatic conditions are appropriate.

6. DISCUSSION

The purpose of this discussion chapter is to place the findings of this study within a framework of restoration success and briefly discuss the management implications of this research.

6.1. Introduction

Ecological restoration is a process of assisting the recovery of a degraded system (SER, 2002). The purpose of ecological restoration is to accelerate successional processes at a degraded site so that a desired community is achieved sooner than would be attained through natural succession (Honnay et al, 2002; Palmer et al, 1997; Bradshaw, 1987), or in some instances to shift a system from one state to a more desired state (Hobbs & Norton, 1996). Restoration of structure and composition without function, or re-creation of ecosystem functioning in the absence of structure and composition, fails to constitute complete restoration (Reay & Norton, 1999a). Observations of both structure and function of the restored ecosystem therefore, are critical to restoration success (Bradshaw, 1987; 1983). Appropriate parameters to monitor for restoration success depend on the goals of a restoration project (Holl & Cairns, 2002). Success is directly related to how effective the restoration effort has been in achieving goals established at the outset of the project.

Suggestions of appropriate key ecosystem attributes to be used as measures for assessing restoration success are prevalent in the literature (e.g. Holl & Cairns, 2002; Hobbs & Norton, 1996; Aronson et al, 1993a; 1993b; Cairns, 1993; Westman, 1991). Aronson et al (1993a; 1993b) define a series of vital ecosystem attributes (or VEAs) that are correlated with and can serve as indicators of ecosystem structure and function at a given developmental stage. Hobbs & Norton (1996) list seven ecosystem attributes to be restored: composition, structure, pattern, heterogeneity, function, dynamics and resilience. Cairns (1991) also suggests several measures of success, including the restoration of ecosystem services such as carbon storage and the restoration of successional processes. Montalvo et al (1997) regard the establishment of successional processes, characterised by species that were not part of the original

planted biotic mix, as an appropriate criteria to judge, in part, the success of restoration.

Measures of success used must relate specifically to a restoration goal (Hobbs & Harris, 2001). Holcim's restoration goal was to establish a mosaic of indigenous forest and wetland communities similar to that which would have existed prior to human (principally European) arrival (Norton, 1992). Implicit in this goal was the desire to restore both ecosystem structure and function so that a self-sustaining and fully functioning ecosystem could develop. A range of structural, compositional and functional measures were used to assist in the determination of the initial success of Holcim's restoration plantings.

The process of restoration involves directing ecosystem development along a desired trajectory (Hobbs & Norton, 1996). Figure 6.1 illustrates a variety of possible trajectories that a system can travel along. The recognition that alternative states are possible in any location, under any conditions is important. It implies that a restoration effort is unlikely to achieve a single desired community (Hobbs & Norton, 1996).

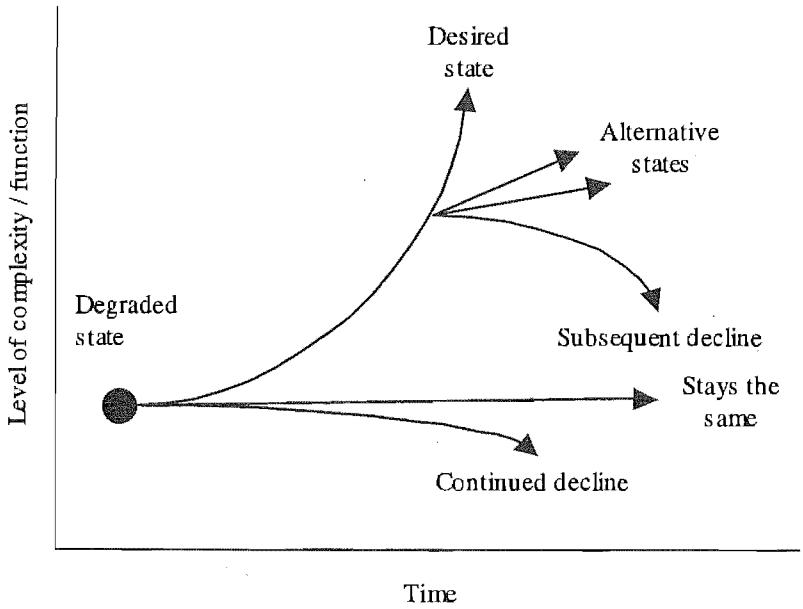
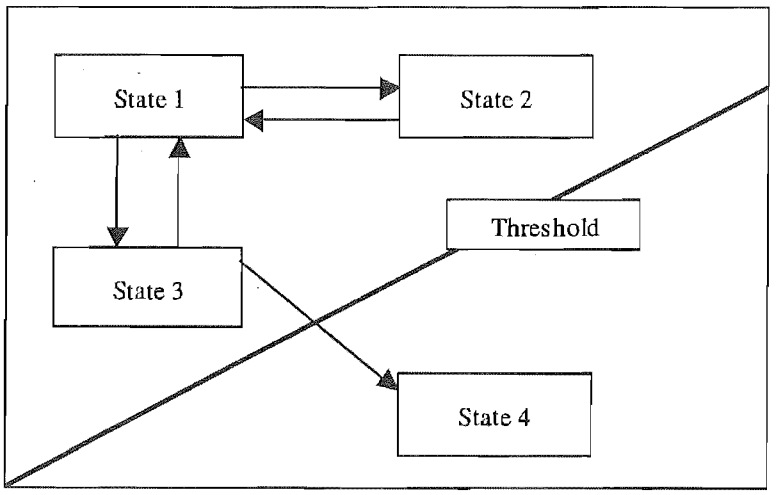


Figure 6.1: A traditional view of restoration of a degraded system, illustrating the idea that the system can travel along a number of different trajectories and that the goal of restoration is to hasten the trajectory towards some desired state. This figure does not consider the history of the system undergoing restoration, despite the implications that site history has for restoration potential. Taken from p.98 of Hobbs & Norton (1996).

Fully functioning systems have natural recovery processes that maintain sustainable flows of soils, nutrients, water and organic materials (Whisenant, 2002). Degrading processes positive feedback mechanisms reinforce and accelerate damaging processes with the potential to result in irreversible vegetation change, once a site's capacity to self-repair has been exceeded (Whisenant, 2002). Contemporary succession theory describes this catastrophic change as having crossed a transition threshold that inhibits natural recovery. This change within damaged ecosystems is unlikely to be an ordered and gradual development (Whisenant, 2002). Such transition thresholds often require massive inputs to restore systems to a condition more similar to their original state (Figure 6.2).

Transitions between states through restoration may be difficult to achieve if they involve changes in composition in terms of functional groups they represent (Hobbs & Norton, 1996). For example, forcing a change from grassland to a shrubland is more difficult to achieve than a change from one type of grassland to another (Hobbs & Norton, 1996). This transition from grassland has been greatly accelerated at Cape Foulwind through the physical planting of shrub species (i.e. planting has forced the transition).

(a)



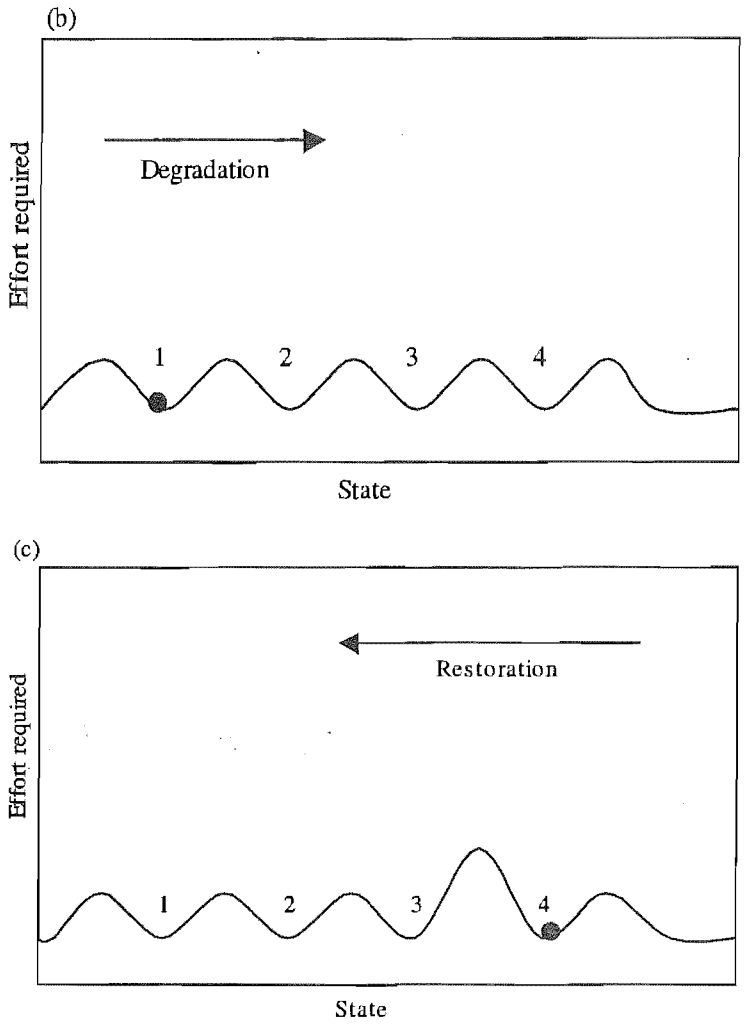


Figure 6.2: A state and transition approach to restoration. Figure (a) depicts a hypothetical system able to exist in four alternative stable states. State 1 is an undegraded state, states 2 and 3 are partially degraded, and state 4 is highly degraded. Transitions from state 1 to other states occur in response to stressors, be they different stressors, or varying levels of the same stressor. Transition back from states 2 and 3 to state 1 are possible if the stressor is removed. However, a transition from state 3 to state 4 involves crossing a threshold that precludes a return to state 3 without increased management, even with the removal of the stressor. Figures (b) and (c) are illustrations of the degree of effort required to force transitions between states. The processes of degradation may force transitions that are much more difficult to force back during the restoration process. Taken from p.99 of Hobbs & Norton (1996).

Two types of threshold barriers (illustrated in Figure 6.3) are thought to inhibit recovery of systems without increased management intervention: biotic and abiotic (Whisenant, 2002). Restoration of degraded systems requires the removal of influences that led to the degradation (Hobbs & Norton, 1996). Biotic interactions form threshold barrier controlled by interference from other organisms, such as invasive weeds, inhibiting natural recovery. Removal of such problematic species through the use of herbicides, mechanical or hand treatments and fire (where

appropriate), are seen as the most effective strategies for such circumstances (Whisenant, 2002). Alternatively, biotic thresholds can result from a loss of mutualists or altered trophic interactions. Often however, systems will not respond directly to the removal of the degrading influence, or stressor, and will necessitate management intervention. Abiotic limitations to recovery are apparent when hydrological processes or harsh microenvironments occur. Physical improvements to the environment are required in these situations (Whisenant, 2002).

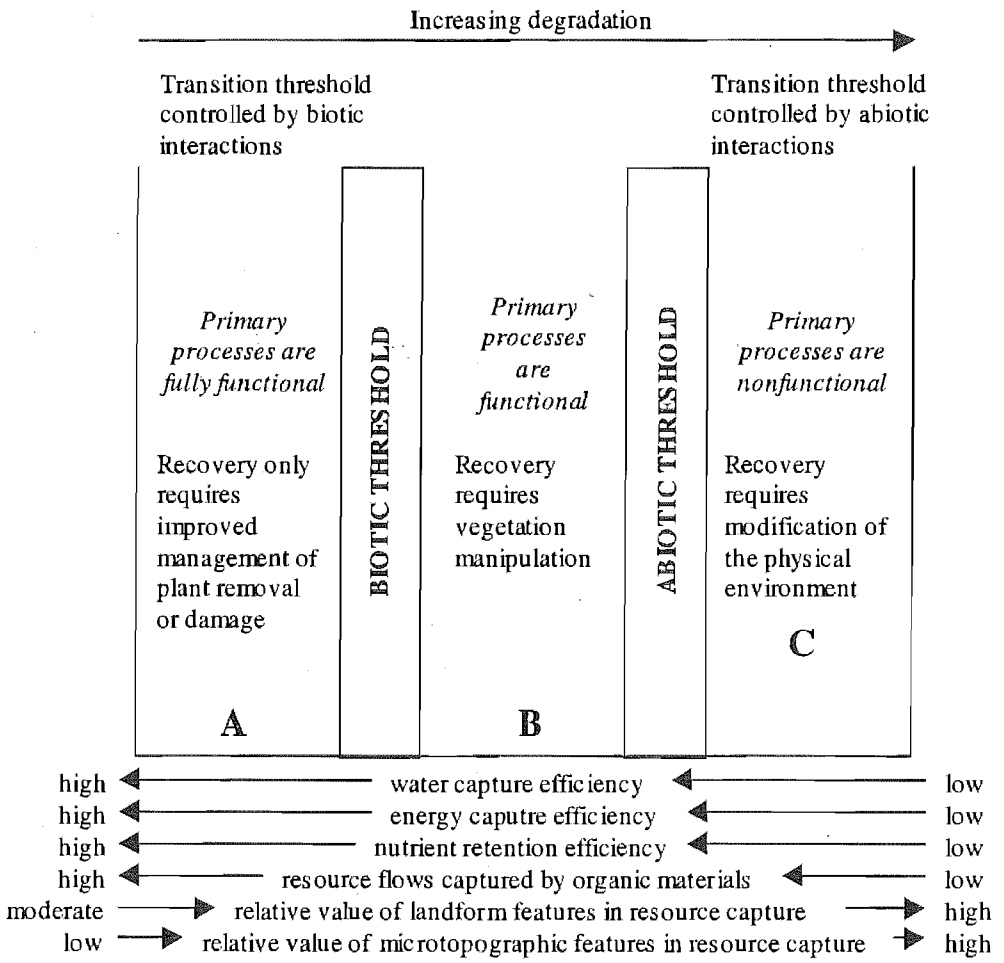


Figure 6.3: Degradation of hypothetical vegetation illustrating the two common transition thresholds that separate three vegetative groups (A, B and C) of functional significance. Taken from p.85 of Whisenant (2002).

The three vegetative groups in Figure 6.3 (A, B, and C) are defined by their functional integrity rather than species composition. Areas A and B illustrate healthy ecosystems that retain limited resources through a combination of biotic and abiotic controls over the resources flowing through the landscape. Biotic flow of resources occurs via both

living and dead organic materials. Area C represents a severely damaged ecosystem, with fewer plants and organic materials (Whisenant, 2002). On the most degraded sites, initial plant establishment may be unable to occur until physical manipulations to the soil surface increase resource availability, allowing plants to establish and begin to exert biotic control over limiting resources.

An understanding of the physical environment, including ecological processes operating at a site, as well as barriers to natural recovery (Whisenant, 2002) is critical to the success of restoration (Montalvo et al, 1997). Healthy ecosystems have natural recovery processes that enable the maintenance of sustainable flows of soil, water and organic materials. Such systems have sustainable resource fluxes, where resource losses are offset by resource gains (Whisenant, 2002).

Reay & Norton (1999a) describe ecological restoration as occurring along a continuum from the successful establishment of initial plantings through to the establishment of attributes that ensure a self-sustaining, fully functioning system (Figure 6.4). The successful attainment of the initial stages of this continuum are indicative of the likely success of the latter stages. They suggest that the use of a continuum along which success can be gauged enables evaluation of success to be achieved with greater ease (Reay & Norton, 1999a).

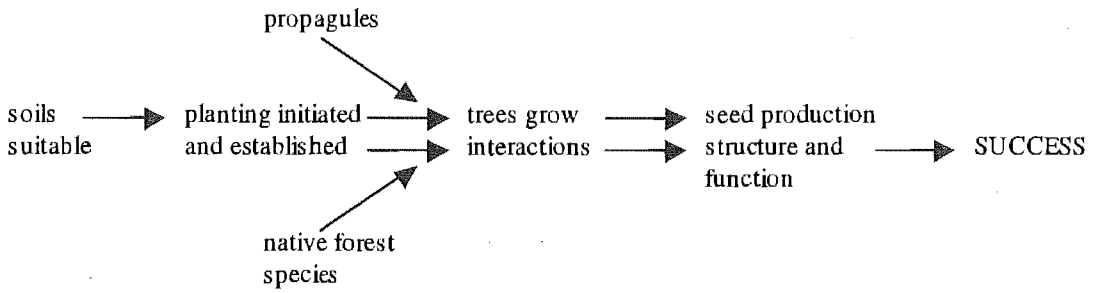


Figure 6.4: Restoration continuum. Taken from p.95 of Reay (1996).

The assessment of the initial success of Holcim's plantings at Cape Foulwind was the overarching goal of this study. More than the successful establishment of plantings, which was regarded as sufficient for initial success by Reay & Norton (1999a) and Reay (1996), an indication of the development of structural and functional attributes necessary for the attainment of a self-sustaining fully functioning system were investigated to suggest initial restoration success in this study.

In order to evaluate the initial success of Holcim's restoration plantings at Cape Foulwind this discussion chapter has been divided into four principal sections. The development of ecosystem structure and function within the planted restoration study sites are dealt with separately in this discussion. Both structure and function are vital ecosystem attributes, which although interrelated, may develop over different temporal scales (Palmer et al, 1997). The first section (Section 6.2.) deals with the question of whether ecosystem structure / composition has been restored. The restoration of ecosystem functioning is dealt with in Section 6.3. The analysis of each of these sections is summarised in the third section (Section 6.4.), which asks whether initial success of restoration plantings at Cape Foulwind has been achieved. A subsection within this (Section 6.4.1.) draws inferences on the possible future development of these restoration plantings. The final section (Section 6.5.) in this chapter briefly discusses the management implications of this study for Holcim.

6.2. Ecosystem Structure

Ecosystem structure, as used within this study, is comprised of two aspects: composition and structure. Composition refers to species presence and their relative abundance, while structure includes aspects of vertical and horizontal patterning and heterogeneity (Hobbs & Norton, 1996).

Numerous parameters have been suggested within the relevant literature as appropriate parameters with which to measure structure. Holl & Cairns (2002) suggest landscape, amount of edge, connectivity, fragmentation, patch size, and proportion surrounded by different habitats as potential parameters for measuring vegetation structure. Also mentioned are species composition, biomass and soil nutrients (Holl & Cairns, 2002). Westman (1991) suggests composition, abundances (absolute and relative), gene frequencies, pattern of local and regional distribution, density, biomass, nutrient pools, topographic features, water quality and quantity, energy content, soil structure and soil / litter nutrient pools as parameters of ecosystem structure (Reay, 1996). Aronson et al (1993a; 1993b) list the following vital ecosystem attributes (VEAs) to be used to indicate ecosystem structure: perennial species richness, annual species richness, total plant cover, soil-borne seed bank, above ground phytomass,

beta diversity, life form spectrum, keystone species (preference or activity), microbial biomass and soil biota diversity.

Attempts to gain detailed understanding of the structure and composition of the restoration plantings and surrounding remnants in the study area were constrained by numerous practical considerations, primarily time available and cost. Although Aronson et al (1993a; 1993b) designed their VEAs to be applicable to all (or nearly all) ecosystems, while retaining sufficient sensitivity to disturbances (human and otherwise), to show variations within a few years (Aronson et al, 1993b), it is recognised that criteria used to judge whether a restoration has been successful need to be specific to the project at hand (Holl & Cairns, 2002; Montalvo et al, 1997). Ecosystem structure was evaluated in this study using vegetation (Chapter 3), ground active invertebrates (Chapter 4) as well as analysis of soil nutrient pools (Chapter 5).

The first objective of this study, outlined in Chapter 1, was to evaluate whether a progression in the development of vascular plant and invertebrate species has occurred with increasing time since planting. This objective is used to shape this ecosystem structure section.

6.2.1. Was a Progression in the Development of Vascular Plant Species Evident with Increasing Time Since Planting?

A clear difference in vegetation composition was evident between planted restoration and remnant study sites. Significantly more woody species were found as part of the canopy stratum in remnant sites than were found in planted restoration sites. There was no clear trend evident for the number of species to increase towards that found in remnant study sites with increasing time since planting. In fact, heterogeneity of the planted restoration study sites was found to progressively decrease with increasing time since planting. It is recognised however, that the number of woody species found in planted restoration sites, excepting regenerating seedlings, was primarily a result of the variety of species planted. Woody species found in the P6 study site was an exception to this as some regeneration had grown to a height at which it was considered part of the canopy (personal observation).

The majority of species planted in planted restoration study sites were not found in remnant sites. *Coprosma propinqua* and *C. robusta* were the only planted species found in the canopy stratum of both planted restoration and remnant study sites. The P6 study site was found to be the most compositionally similar, of all the planted restoration sites, to a remnant site (R3), although this similarity was still. Planted restoration study sites will only become compositionally similar to remnant sites once the planted restoration component becomes markedly less dominant, provided species colonising the planted sites are the same as those found in remnant study sites. The results of a study by Reay & Norton (1999a) suggest that such development could take at least 30 years (compared with the 22 years since the 'oldest' restoration site studied at Cape Foulwind was planted).

Despite the limited similarity between remnant and planted restoration study sites, and the lack of a progression for this similarity to increase with time since planting, the presence of regenerating seedlings of species that were not planted indicates that this is likely to change in the future. The likely development of the restoration plantings is discussed in Section 6.4.1.

Indications of vegetation structure can be found within diversity measures. Mean cover and heterogeneity of woody species were significantly greater in remnant sites than in planted restoration sites. Further, cover of woody species tended to increase with time since planting for the first three planted restoration sites (P1, P2, and P3). Much variation was encountered around the mean values of all diversity measures implying that results should be interpreted with a degree of caution. Heterogeneity progressively decreased with increasing time since planting implying that planting has occurred in a mixed fashion (as opposed to block plantings) within more recently planted study sites.

Variation in vegetation structure was not entirely evident from the diversity indices alone. Woody species were present within a shrub stratum only in planted restoration study sites. Woody vegetation in R1 and R2 study sites was spread throughout three canopy strata: shrub, sub-canopy, and canopy. Vegetation in R3 was found in both shrub and canopy strata. Estimated values of increased height in planted restoration study sites illustrated that vegetation height increased generally with increasing time

since planting. The growth form of the planted species does, however, have a strong influence on canopy height in the restoration plantings. Nonetheless, such progressive height increases with time since planting does indicate the tendency of canopy height to increase toward that of remnant study sites.

6.2.2. Was a Progression in the Development of Ground Active Invertebrates Evident with Increasing Time Since Planting?

The abundance and composition for ground active invertebrates were found to differ between planted restoration and remnant study sites. The ordination diagram of 'all invertebrates' (Figure 4.13) indicated a high degree of similarity between invertebrate composition within remnant study sites. The same diagram however, suggested a larger degree of disparity between invertebrate composition in planted restoration study sites. More species of ground active invertebrates were caught in pitfall traps in planted restoration sites, however this number was not found to be significantly different than the number of invertebrates caught in remnant study sites. Some species caught in planted restoration sites may have differed because they were grassland species, compared with forest dwelling species trapped in remnant study sites. The number of individuals trapped in remnant sites was found to be significantly greater than that found in planted restoration study sites. Simmonds et al (1994) suggest that microhabitats within forest sites may not provide suitable conditions for certain species of spiders, particularly opportunistic pioneer species. The various stages of the planted restoration study sites may provide opportunities for this group of spiders to exist.

Studies by Simmonds et al (1994) and Majer & Nichols (1998) measuring the composition of spider and ant communities respectively, in order to assess the success of restoration of mine sites, both reported trends in the development of invertebrate community composition associated with habitat development (a function of site age). Developmental trends in both studies were determined by correlating environmental variables such as plant species richness and density, leaf litter depth and percentage cover, soil compaction with the number of invertebrates caught. Planted restoration study sites in this study were found to contain a significantly more heterogeneous array of invertebrates than remnant sites. However, no clear trend for change in species richness of invertebrates trapped was evident with increasing age of planted

restoration study sites. This was probably due to the low number of invertebrate species caught in the P6 study site.

Patterns found in the composition and abundance of 'all invertebrates' caught were similarly displayed through analysis of Coleoptera caught. Ground active invertebrates are ubiquitous, performing a variety of functional roles (Hutcheson & Kimberley, 1999). They dominate the functions and processes of most ecosystems (Majer et al, 2002; Keesing & Wratten, 1998) and are important indicators of ecosystem functioning (Webb, 1996). Ants (e.g. King et al, 1998; Andersen & Sparling, 1999; Andersen, 1993), spiders (Simmonds et al, 1994) and Coleoptera (Abildsnes & Tømmerås, 2000; Hutcheson 1996; 1990) have all been suggested as suitable environmental indicators of ecosystem function. Coleoptera species were significantly more abundant in remnant than planted restoration study sites. The presence and relative abundance of food sources for Coleoptera may have played a role in influencing their distribution. Predatory Coleoptera species dominated planted restoration study sites, both in terms of species and the number of individuals caught. While herbivorous species of Coleoptera dominated remnant study sites, both in terms of species and the number of individuals caught. Dense vegetation should provide a greater food source for herbivorous species, and higher levels of detritus (Wheater & Cullen, 1997), accounting for the greater proportion of herbivorous Coleoptera detected in remnant study sites.

The pattern of Araneida distribution throughout the study area was a little less clear to decipher, probably due to the smaller numbers collected. A less obvious distinction was apparent in the ordination diagram (Figure 4.17). Significantly more individuals of Araneida were caught in planted restoration sites than in remnant study sites. The greater number of Araneida individuals found in planted restoration sites may be the result of the comparatively open habitats allowing for habitation by the predominantly opportunistic Araneida species (Morris, 2000). No significant difference was found in any diversity index between the six planted restoration study sites. P4 and P5 were the study sites that shared the greatest degree of similarity in terms of Araneida species caught.

6.2.3. Did Soil Nutrient Status Vary Between Study Sites?

Ecosystem development relies on adequate quantities of nutrients available in the soil (Bradshaw, 1983). All soil samples collected were acidic indicating that conditions suitable for accumulation of soil organic matter in planted restoration study sites are developing (Honnay et al, 2000). Nutrients were primarily found in greater quantities in remnant study sites than in planted restoration sites. Total carbon, nitrogen and potassium were all found in significantly greater quantities in soil samples from remnant sites.

No clear progression of soil nutrient status was evident with increasing time since planting. The P5 study site had the lowest levels of total carbon and nitrogen. P1 contained the highest percentage of total carbon, while soil samples from P2 were found to have the highest levels of total nitrogen. The variation in available nutrients would depend to some degree on the amount of fertiliser applied during planting and during land management prior to planting. An influential factor in determining the soil nutrient status of soils within the study area was the type of substrate. The P1, P2 and P6 study sites were all situated on overburden dumps. The variation in substrate types and fertilisation regimes post-planting, may have obscured results.

6.3. Ecosystem Function

Ecosystem function must be restored to the planted restoration study sites in order for the restoration to be considered successful. Restoring the planted restoration study sites so that they become self-sustaining, fully functioning systems requires understanding of ecosystem functioning. A properly functioning system is one that has sufficient resilience to persist despite natural environmental fluctuations (Holmes & Richardson, 1999; Palmer et al, 1997).

The re-establishment of a vegetation assemblage is not sufficient to restore a fully functioning system (Majer et al, 2002). Hobbs & Norton (1996) suggest ecosystem function involves the performance of basic ecological processes such as energy, water, and nutrient transfer. Majer et al (2002) list functions such as nutrient retention and cycling, purification of air and water, detoxification and decomposition of wastes, pollination, and seed dispersal as necessary components of a fully functioning system.

In order to achieve restoration success, the re-establishment of ecosystem processes such as seed dispersal and parasitism are essential (Norton, 1995).

Aronson et al (1993a; 1993b) list eleven vital ecosystem attributes to serve as indicators of ecosystem function when assessing restoration success. These include biomass productivity, soil organic matter, soil surface conditions, coefficient of rainfall infiltration, maximum available soil water reserves, rain use efficiency, cation exchange capacity, length of water availability period, nitrogen use efficiency, microsymbiont effectiveness, and cycling indices (Aronson, 1993a; 1993b). Westman (1991) suggested productivity / growth rates, nutrient flux, pollutant flux, natality / mortality rates, migration, fire frequency / intensity, hydrological flow, soil movement and radiation flux as parameters to be used as indicators of ecosystem function.

Structural attributes are often used as indicators of ecosystem function due to their relative ease of assessment (Chambers et al, 1994). Structural and functional components of ecosystem however, do not necessarily establish or recover at identical rates (Holl & Cairns, 2002). Not all structural and functional measurements are correlated with each other, however they are connected through complex interrelationships (Holl & Cairns, 2002). This emphasises the importance of using a range of ecosystem parameters to monitor restoration success.

Ecosystem process may impact the study area over a range of spatial and temporal scales. In addition to the size or duration of ecosystem processes, they may impact only a portion of the study area, regardless of size or duration (Parker, 1997). For example, topographic variation modifies environmental factors such as temperature. Directional processes such as salt spray from the ocean may differentially impact study sites dependent on their exposure to onshore winds (Parker, 1997). Further, the extent of the impact of processes is dependent on the context of the study area. For example, the size and shape of the nine study sites, in addition to the surrounding habitat and overall landscape mosaic will affect the extent of impact of ecosystem processes (Parker, 1997).

The necessity of restoring all components of biodiversity in order to restore ecosystem function has been debated widely (e.g. Majer et al, 2002; Holmes & Richardson,

1999; Palmer et al, 1997). It is unlikely that each species within a system forms a separate and indispensable link in processes involved with any measurable function. Many species play a disproportionate role in ecosystem function (Palmer et al, 1997). This leads to an argument regarding species redundancy (Holmes & Richardson, 1999). It is incorrect to assume that species are functionally analogous, thereby do not need to be replaced in a restoration project. The maintenance of a fully functioning ecosystem may not depend on restoring entirely the full range of species, but on the species restored being dynamic enough to ensure a capacity for resilience to particular disturbance events (Holmes & Richardson, 1999). Increased diversity stabilises the functioning of ecosystems by increasing this resilience to perturbations (Majer et al, 2002). In restoration however, there is a need for pragmatism, accepting that restoration of all species will not typically be possible (Palmer et al, 1997).

As is apparent from the above discussion, numerous parameters for measuring ecosystem function are available. Whilst measurement of all such parameters was not practical, evidence of a range of parameters was necessary to assess the initial success of Holcim's restoration plantings at Cape Foulwind. The choice of parameters of the purposes of this study were constrained by practical issues such as time and cost, in addition to the availability of equipment. Therefore a range of parameters, both structural and functional, were used to assess restoration success. Parameters were chosen that would provide the greatest amount of information within the constraints of this study. Through the comparison of planted restoration with native remnant study sites indicators of ecosystem function could be assessed with greater ease (Chambers et al, 1994). The need to focus on dispersal and colonisation dynamics was apparent (Palmer et al, 1997).

Evidence of regeneration in five of the six planted restoration study plots indicated that functional processes necessary for the colonisation and germination of seedlings were present. Regenerating seedlings were only absent from the P1 study sites, which had only been planted three years prior to this study. Seedlings found in P5 were from species present in the canopy of this site, indicating that pollination was occurring. Study sites P2, P3, P4, and P6 contained evidence of regenerating seedlings of species present in the shrub layer of these sites, in addition, species not present as part of the

canopy were also present (i.e. dispersal of novel species into these four study sites had occurred).

Woody species have strong ameliorating effects on the microenvironment (Whisenant, 2002). Such effects result through passive means, such as the effect of their physical structure shading the soil and altering wind movements resulting in increased relative humidity, or through active means such as trapping soil, nutrients and propagules of both microorganisms and other plant species. Metabolic processes alter temperature, humidity as well as the physical and chemical processes of soils. Increases in soil organic carbon and improvements in the water and nutrient holding capacity of soils are additional consequences of vegetation establishment (Whisenant, 2002). The presence of regeneration within five planted restoration study sites suggests that microhabitat conditions suitable for germination were present, such as a litter layer and appropriate light and moisture conditions. Regeneration density was found to increase proportionally to an increasing litter depth and volume (Figure 5.4, Figure 5.6). The presence of species such as *Dacrycarpus dacrydioides* also shows that the planted restoration sites are facilitating dispersal, as this species is dependent upon bird dispersal to reach these sites.

Litter accumulation is an important process leading to the development of soil organic matter and increased nutrient retention while reducing erosion (Whisenant, 2002). Litter development did not appear to follow strictly to the age of the planted restoration sites. Although no evidence of ground litter was found in the 'youngest' study site (P1) and P6 contained the deepest litter layer (significantly deeper than P1, P2, P4 and P5), a clear progression with age was not apparent for the other four study sites. The presence of ground litter appeared to be affected by the pervasive grass sward. Ground litter was primarily found directly under planted species only in these planted restoration study plots. Litter depth, of the environmental variables investigated, was found to be the key driver of the distribution of ground active invertebrates over the study area, supporting the role of litter as a key factor in ecosystem development.

The second objective for this study listed in Section 1.5. was to determine whether the ecosystem processes of litter decomposition and seed dispersal have been established

within the planted restoration study sites. This objective is broached to assist the discussion regarding ecosystem functioning in planted restoration study sites.

6.3.1. Was Litter Decomposition Occurring Within Planted Restoration Study Sites?

Study of short term litter dynamics within all study sites revealed that litter decomposition was indeed occurring, and that the amount of decomposition did not differ significantly between remnant and planted restoration sites after three, six, or nine months in the field. Initial mass loss of *Coprosma robusta* from litter bags occurred quickly. The greatest loss occurred within the initial three month period in the field; all litter bags lost more than 50% of their original estimated dry weight; there were no significant differences between sites.

6.3.2. Was Seed Dispersal Occurring in Planted Restoration Study Sites?

The context of the study area is an important factor affecting the degree to which restoration success can occur (Palmer et al, 1997; Parker, 1997; McClanahan & Wolfe, 1993). The planted restoration study sites are all in close proximity to remnant study sites, providing a readily available source of seeds to be disseminated into the planted sites.

Seed traps in remnant study sites contained significantly more woody species than those established in planted restoration sites. Seed deposition is limited by the availability of suitable perch sites (Ferguson & Drake, 1999). The higher number of woody species in remnant seed traps may have resulted from preferential perching by birds at a particular location. Alternatively, the position of seed traps near to, or directly underneath some species may have lead to seeds falling directly off trees into seed traps. No significant difference was found between the number of seeds of woody species in seed traps within planted restoration study sites. One sixth of woody species caught in seed traps within planted restoration study sites were novel species (i.e. species that were not present within the shrub stratum of planted restoration sites), suggesting that active dispersal is occurring. The presence of novel species (e.g. *Dacrycarpus dacrydioides*) in the seed rain collected within planted restoration sites is evidence of successful dispersal within these study sites.

6.4. Has Initial Restoration Success Been Achieved?

The overarching goal of this study was to assess the initial success of the restoration plantings at the Holcim quarry based at Cape Foulwind, New Zealand. In order to achieve this aim an investigation into various aspects of the development of the restoration plantings was undertaken. Ecosystem parameters investigated included vegetation composition, ground active invertebrate composition and abundance, soil nutrient status, seed rain, and litter dynamics. The results of this study suggest that while complete success of these restoration plantings has not yet occurred, initial success has indeed been achieved, as shown by the presence of a developmental progression towards remnant study sites evident with increasing age of planted restoration sites.

The planted restoration sites are yet to be restored to be structurally or compositionally similar to the reference remnant sites. Achieving ecosystem structure that is identical to remnant sites is neither feasible nor desirable. Whisenant (2002) suggests that initial restoration efforts should focus not only on establishing species, but that species used should initiate processes that enhance ecosystem functioning. Such processes are being facilitated by the current restoration plantings. Planted restoration sites are progressing and facilitating the entry of succession species. This implies that the initial stages necessary for a progression to a self-sustaining ecosystem are being provided for. The current limiting factor to progression within the planted restoration study sites appears to be the lack of full canopy cover, which will develop with time.

The restoration plantings have been successful in terms of providing new habitat for native biota. The vegetation present in planted restoration study sites appears sufficient to attract bird species that disperse novel species into these areas, thereby facilitating regeneration of woody species and accelerating ecological succession of these plantings. Dispersal does not guarantee the establishment of desired species even if soil and microclimatic conditions are appropriate. The establishment of a full canopy cover appears to be the key driver in the further development of these restoration plantings. Once this has occurred, suitable microhabitat conditions should develop.

A large difference was apparent in composition and abundance of ground active invertebrate communities in planted restoration and remnant study sites. This difference was particularly evident for the Coleoptera category. Coleoptera collected from both types of study site revealed a difference in functional diversity. Greater numbers of herbivorous Coleoptera were located within remnant sites, while predatory Coleoptera dominated planted restoration study sites. This dominance of predatory Coleoptera within planted restoration study sites was thought to be a consequence of a paucity of ground litter within these sites, and hence a lack of appropriate niche sites. Dense vegetation should provide a greater food source for herbivorous species, and higher levels of detritus (Wheater & Cullen, 1997), possibly accounting for the greater proportion of herbivorous Coleoptera detected in remnant study sites. Litter depth, of the environmental variables investigated, was found to be the key driver of invertebrate distribution over the nine study sites.

Litter decomposition was able to occur within all study sites. A lack of litter within the P1 study site and limited presence of ground litter within other planted restoration sites would limit the occurrence of litter decomposition under normal circumstances. The occurrence of such an ecosystem process is valuable, as decomposition is an essential component of nutrient cycling, again suggesting the planted restoration sites are progressing towards becoming fully functioning systems.

The results of this study suggest that components and functions deemed necessary for initial restoration success are present. Holcim's restoration plantings at Cape Foulwind have successfully provided new habitat for native biodiversity. Importantly, they are increasing the connectivity between the native forest remnants, enhancing the continuity of the site and further enhancing the overall aesthetic appeal of the area.

As stated at the beginning of this chapter however, attainment of restoration success depends on the goals established for the restoration project. The specificity, appropriateness, and ease of measurement of these goals will play a large part in determining the level to which restoration plantings can be deemed successful. The overarching goal for Holcim's restoration effort – to restore a mosaic of indigenous forest and wetland communities similar to that which would have existed prior to human (principally European) arrival (Norton, 1992) – is general, and may make

ultimate judgement of success difficult. Suggestions of more specific restoration goals can be seen in Section 6.5.

6.4.1. What is the Likely Future Development of the Planted Restoration Study Sites?

The probability that the planted restoration sites will develop towards a desired state is dependent upon the possibility of colonisation of species from the remnant study sites (Honnay et al, 2002). Colonisation consists of the dispersal of a propagule, which is limited by spatial factors (i.e. the degree to which planted restoration sites are isolated from remnant sites), and seedling establishment and recruitment, which may be constrained by abiotic conditions in the planted restoration study sites (Honnay et al, 2002).

If the assumption that species currently present as regenerating seedlings will establish to dominate the canopy of planted restoration sties in the future, the identity and abundance of this regeneration may be used to provide some indication of the future development of these planted restoration study sites. Regeneration found within remnant sites was predominantly of species identified within part of the existing canopy. This suggests that these remnant sites are self-regenerating and therefore are appropriate systems for which to use as models for judging restoration success (Reay, 1996). It needs to be recognised that the remnant sites included as reference sites in this study, have themselves been subject to disturbances through logging and farming practices.

Isolation, size, shape, resource quality / quantity, and competitive interactions of the study sites may act as 'hurdles' to successful invertebrate colonisation (Keesing & Wratten, 1998). The majority of invertebrates have excellent powers of dispersal, others however, may be limited in their capacity to colonise newly restored sites (Majer et al, 2002). Despite the potential to speed up the colonisation of study sites by introducing invertebrate species, invertebrate diversity may be better served by creating appropriate habitat conditions (Majer et al, 2002).

Seedlings detected within planted restoration study sites with the greatest densities, such as *Dacrycarpus dacrydioides*, *Carpodetus serratus*, and *Coprosma grandifolia*,

had not been planted, indicating that dispersal of novel species into planted restoration study sites is occurring. The detection of novel species in planted restoration study sites, which form part of the vegetative structure in remnant sites led to the implication that the planted sites will continue to develop toward the structure and functioning of the neighbouring remnant sites.

6.5. Management Implications

6.5.1. Goals

The original restoration plan prepared for Holcim New Zealand Ltd. (Norton, 1992) outlined a broad restoration goal, or vision, of restoring a mosaic of indigenous forest and wetland communities similar to that which would have existed prior to human (principally European) arrival (Norton, 1992). Such a goal is expansive, and greatly complicates the judgement of restoration success. Restoration success depends on the arrival at mutually agreed upon goals via an open and effective process (Hobbs & Harris, 2001; Higgs, 1997).

More specific restoration goals need to be established and clearly stated. Implicit within this is the need for acceptable levels of variation within ecosystem parameters (Holl & Cairns, 2002). For instance, how similar do the planted restoration sites need to be to remnant sites before they can be deemed successful? Without clearly articulated goals monitoring and judgement of restoration success of the restoration plantings may be difficult (Holl & Cairns, 2002). The assessment of initial success undertaken in this study provides a platform from which future comparisons can be made. A suggestion of appropriate goals has been offered in order to assist future assessment of the success of restoration plantings at Cape Foulwind.

Vision

This long-term vision of Holcim's restoration effort has been adapted from Norton (1992). The restoration will aim to restore a mosaic of indigenous forest and wetland communities, similar to that which would have occurred at the site prior to human (principally European) settlement of the Westport area. The excavation process requisite to quarry operations has resulted in a substantial pit, which will be filled to

form a lake after the completion of mining. Wetland areas will be created around the lake margin, grading back into the forest.

The focus of the following goals is limited to goals appropriate for assessing the success of the restoration of the coastal (zone 1) and inland restoration zones (zone 4) as defined by Norton (1992), as these areas formed the focus of this study. A series of long-term and short-term goals have been suggested.

Long-term Goals (50 year)

1. Connectivity between plantings will have been established, linking directly with remnant sites where possible.
2. The area is being used for recreational, educational and scientific purposes (provided mining operations have ceased).
3. The restoration plantings have been linked to the seal colony and walkway tourist attraction at Cape Foulwind.
4. Direct human intervention is no longer necessary to ensure the continued development of the restoration plantings.

Short-term Goals (10 year)

1. The ecological integrity of the existing remnants and of the restoration plantings has been secured.
2. Restoration planting has continued with at least five additional areas planted.
3. The remaining areas of indigenous vegetation (i.e. remnant sites) have been enhanced and are fully integrated with the restoration plantings.
4. Restoration plantings are growing vigorously; with the establishment of strategically located enrichment plantings.
5. Restoration plantings are facilitating regeneration of species that are prevalent within the remnant areas.
6. Native bird and insect species are prevalent throughout the plantings.
7. The quarry area is kept free of high priority animal pests, while other animal pests are controlled to levels that do not threaten the restoration or additional values of the area.
8. Plant pests are controlled to levels that do not threaten restoration or additional values of the quarry area.

9. A monitoring programme has been established that enables the success of the restoration effort to be quantitatively assessed. Information gathered during this monitoring programme is fed back to Holcim, enabling additional knowledge to be incorporated into future management decisions.
10. The community, particularly locals, but also tourist visitors, are well informed about the restoration project.

6.5.2. Monitoring

Restoration is an ongoing process, ideally resulting in self-sustaining, dynamic systems. Encapsulating this view, monitoring is essential to restoration success (Holl & Cairns, 2002). Monitoring can aid the determination as to whether specific endpoints have been reached. Ideally, baseline monitoring would have occurred prior to the initiation the disturbance, or at least prior to the commencement of restoration efforts. Such baseline measurements provide valuable comparisons when judging restoration success by determining whether change has occurred over time (Holl & Cairns, 2002). In addition to goals being established, monitoring of these goals needs to occur (Holmes & Richardson, 1999). Such monitoring will enable recognition of restoration success to be more readily achieved by providing benchmarks for evaluation (Jackson et al, 1995). This study provides future monitoring efforts at Holcim's Cape Foulwind quarry with a suitable baseline data set that will aid managers to determine the amount and type of intervention necessary as well as decide upon the point at which the restoration effort has ultimately been successful (de Gruchy et al, 2001).

Comparison of planted restoration sites with remnant sites assisted with the determination of initial restoration success. Use of native reference systems exhibiting desired ecosystem properties is a common aid used to determine restoration success (Chambers et al, 1994). However, the selection and role of reference sites has been debated (de Gruchy et al, 2001). Reference systems are used to guide the restoration process with the intention of emulating their structure, functioning, diversity and dynamics (Aronson et al, 1993a). By comparing planted restoration with healthy remnant sites, insights into ecosystem development with implications for ecological restoration can be obtained (Whisenant 2002). Honnay et al (2002) view the use of a reference system as essential to the evaluation of the extent to which restoration goals

have been achieved. Three remnant study sites were used as reference systems in this study. By using three reference sites, natural variation within study sites could be accounted for (Holl & Cairns, 2002).

When the establishment of a self-sustaining system is a goal of this restoration effort, long-term monitoring is essential. Monitoring for only a short period of time is insufficient to determine whether the study sites are self-sustaining (Holl & Cairns, 2002). In order for the restoration plantings to be viewed as self-sustaining, structural and functional attributes need to persist in the absence of intervention (Holl & Cairns, 2002). 48% of restoration projects reviewed by Lockwood & Pimm (1999) ceased monitoring prior to the achievement of goals (Holl & Cairns, 1999).

Ecosystem processes occur at multiple levels. They are dynamic in the spatial and temporal features; therefore, a dynamic approach to monitoring restoration success is essential (Parker, 1997). Long-term monitoring is optimal when considering the success of a restoration project (Michener, 1997). Many questions relating to ecosystem structure and function can only be addressed through repeated sampling of parameters over time. Unfortunately, long-term studies are rare in ecology (Michener, 1997). Trade-offs usually occur within personnel and budget constraints (Holl & Cairns, 2002).

Long-term regular sampling would involve sampling of planted restoration and control sites over a long time period. For example, Majer & Nichols (1998) tracked successional changes in ant communities over 14 years by repeatedly sampling the same three mine pits and a forest control. Through the use of this long-term monitoring approach they were able to conclude that ant communities were converging towards the forest site as time since restoration increased (Majer et al, 2002).

An alternative to this approach is the chronosequence approach, in which multiple sites of differing time since restoration (as used in this study) are used to infer developmental patterns over time. A disadvantage with this approach is that it is assumed that age is the only variable differing between the study sites. This approach is therefore, only valid when the four criteria set out by (Majer et al, 2002) are met:

(1) all sites were the same prior to disturbance, (2) the same magnitude of disturbance was applied to the disturbed sites, (3) all treatments applied to the sites after the disturbance were the same (e.g. topsoil application procedures, planting patterns), and (4) all disturbed sites follow the same pattern or trajectory of recovery (Majer et al 2002). As these four criteria are rarely met, long term monitoring provides the optimal monitoring approach. Whatever approach to monitoring is taken, it is essential that a feedback loop be established so that monitoring efforts such as this inform subsequent management decisions. Permanent plots established as a consequence of this study form the basis for monitoring.

6.5.3. Weed and Pest Impacts

Invasive grasses can restrict the ability of the restoration efforts to achieve success (Robinson & Handel, 1993). It is apparent that fairly intensive post-planting management is necessary to ensure the successful establishment of the planted species and their ability to flourish within such a competitive environment. The invasive grass sward so prevalent within planted restoration study sites plays an influential role in the ability of planted species to flourish, impeding the growth of planted species (see Section 3.4.5.). Once a full canopy cover has been established within the planted restoration sites, these sites will be far less susceptible to invasion by exotic species (exotic grasses and *Ulex europaeus* particularly) (Berger, 1993). *Ulex europeaus* has beneficial attributes, including nitrogen fixation and the provision of a sheltered environment for the establishment of native species (Wilson, 1990). However, the species is likely to slow the restoration process overall (Norton et al, 2001).

Possums and rabbits, which are both recognised as present in the study area, have a major effect on flora and can severely inhibit natural regeneration and the growth of natural restoration plantings. A detailed management plan for dealing with such invasive animal species, which was beyond the scope of this research, would make a valuable contribution to future management of the restoration plantings.

6.5.4. Species Choice for Plantings

Species choice for restoration plantings represents a balance between those species that will (1) achieve optimal growth under prevailing environmental conditions, (2)

those likely to contribute most to meeting the restoration goal, and (3) that will be most attractive to seed dispersing birds (Norton et al, 2001).

At a local level, species choice needs to consider the principal limitations to plant growth (moisture, frost, exposure and salinity, infertility and competition) associated with particular microhabitats (Norton et al, 2001). This should be guided by the success of the restoration plantings to date. Norton (1992) provides a list of species suitable for plantings. However, final species choice needs to be regularly reviewed based on the performance of plantings, and availability of propagated material.

Davy (2002) emphasised that the proportion and planting pattern of species used should reflect their spatial distributions in the target communities. In this restoration effort however, the planted species were intended as nurse species and will not necessarily be found in the desired ecosystem. There is a tendency for the pattern of plantings to give an unnatural appearance for the first few years, due to a regular planting pattern. A small-scale, irregular pattern that is repeated across sites may provide the optimal approach for a 'natural' appearance in the initial stages of vegetation growth (Davy, 2002).

There may be species that are particularly successful 'nurse species', creating environmental conditions that facilitate the establishment and growth of additional indigenous species. For example, *Phormium tenax* has been found to be a particularly successful nurse species, providing appropriate microclimatic conditions that were chemically and physically suitable for the regeneration of desired native species on the Port Hills, Canterbury (Reay & Norton, 1999b). This observation was not substantiated within this study however, although providing a potentially appealing perch site for birds, no regeneration was observed within *Phormium tenax* clumps. Numerous species used in the planted restoration sites, particularly *Coprosma robusta* and *Pittosporum eugenioides* and *Pittosporum tenuifolium*, were found to be satisfactory nurse species, attracting frugivorous birds bearing seeds, which were then able to establish as seedlings. Therefore, it is evident that the planted restoration study sites are facilitating the process of establishment of woody species (Webb, 1996), although the extent to which this can occur is currently limited to by the vigorous grass sward.

Simpson (1992) and Timmins & Wassilieff (1984) both emphasise the importance of planting in restoration projects using only local provenances (eco-sourcing) in order to maintain the genetic integrity of a restoration site. Due to adaptation of vegetation to particular conditions, the use of genetically appropriate species may greatly affect the level of restoration success achieved (Montalvo et al, 1997).

An additional lesson learnt during the establishment of plantings in this area is the importance of the appropriateness of species used in restoration (Harris, 1997; Simpson, 1992). *Coprosma propinqua* was planted widely throughout this area, particularly in exposed locations. Despite this species being present naturally in the wider area it has struggled to establish in the presence of vigorous exotic grass growth, due to much of its growth occurring laterally (Keir, 1998). Keir (1998) notes that despite *C. propinqua* being suited to the climatic conditions of the study area, its slow growth, primarily produced low down and laterally, it has struggled to grow within the dense grass sward. Due to the difficulty establishing, hand releasing of individual plants was necessary. Such intensive post-planting management is less than desirable. As recommended by Keir (1998), this species should be used in the future only where hand releasing is undemanding and pasture growth is limited.

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